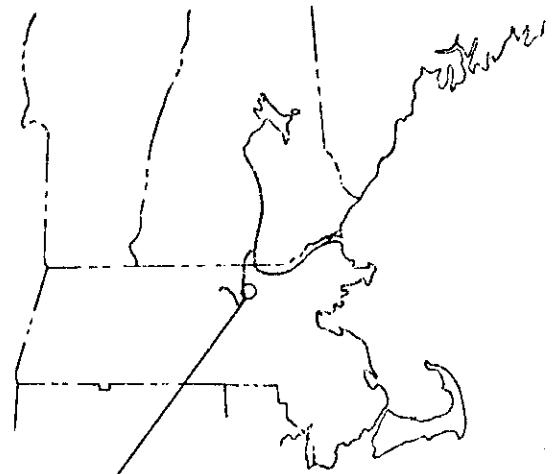
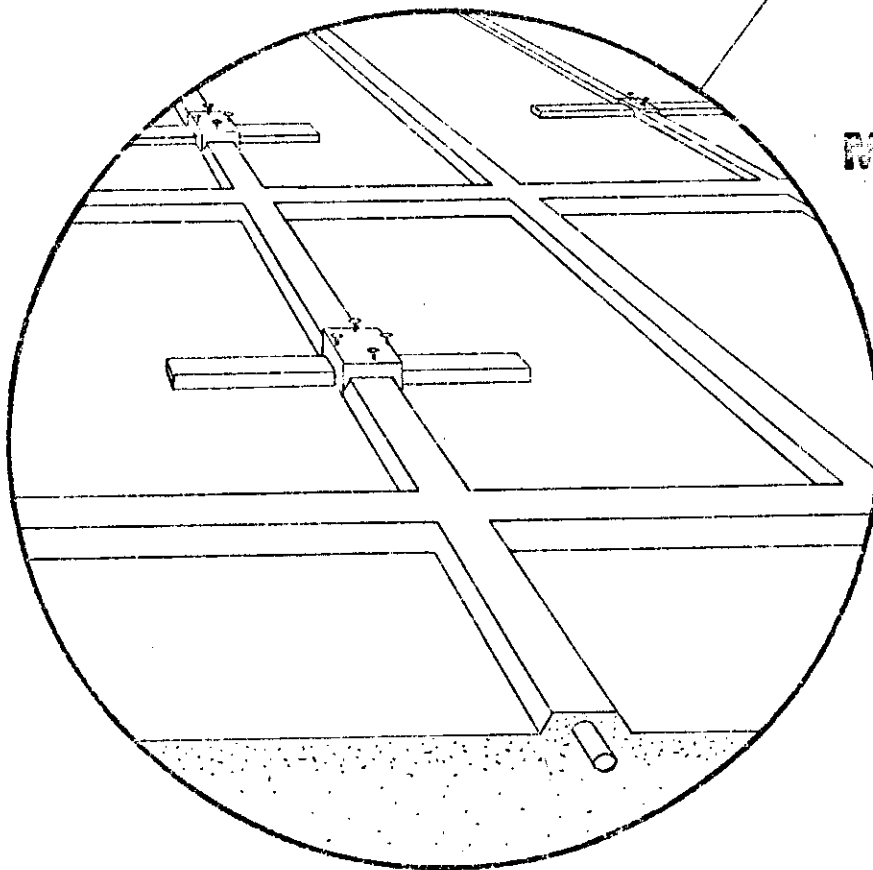


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# **RAPID INFILTRATION OF PRIMARY SEWAGE EFFLUENT**



**FORT DEVENS,  
MASSACHUSETTS**



RAPID INFILTRATION OF PRIMARY SEWAGE EFFLUENT

at

FORT DEVENS, MASSACHUSETTS

by

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## ABSTRACT

Rapid infiltration has provided final treatment to unchlorinated Imhoff tank effluent at Ft. Devens, Massachusetts since 1942. Wastewater flow has varied seasonally; however, most flows to the 22 treatment beds in 1973 were in the range of 2676-9541 m<sup>3</sup>/d (0.7-2.6MGD) with a mean daily flow volume of 5049 m<sup>3</sup>/d (1.3MGD). Adhering to an operation cycle entailing simultaneously inundating three 0.31 hectare treatment beds for two days followed by 14 days recovery period effluent application has been about 27.1 m/yr.

Chemical analysis of soil samples obtained from the upper 3.05 m of the treatment beds showed that organic matter, conductivity, nitrogen, phosphorus, calcium, magnesium, potassium, manganese, sodium, copper, iron, and zinc were substantially higher than background levels, while sulfur, chloride and cation exchange capacity were only slightly higher. Treatment bed soil samples had lower pH levels and boron concentrations than levels found in background samples.

Quality of the primary effluent applied to the treatment beds and groundwater in fourteen observation wells was determined through sampling at biweekly intervals and analysis of samples for pH, conductivity, alkalinity, hardness, five day biochemical oxygen demand, chemical oxygen demand, total nitrogen, organic nitrogen, ammonia-N, nitrate-N, nitrite-N, total phosphorus, ortho-phosphorus, chloride, sulfate and total coliform bacteria. Groundwater quality in wells located 60-100 from the application area showed the primary effluent after flowing through the sand and gravel formation had received substantial renovation. Prior to treatment, effluent COD and BOD<sub>5</sub> levels in the groundwater ranged 10-20mg/l and 1.0-2.5 mg/l. Concentration of total nitrogen in the primary effluent was 47 mg/l which was equally organic and ammonium nitrogen. Total nitrogen in water samples from observation wells affected most was 10-20 mg/l, primarily as nitrate-nitrogen. This was a reduction in total nitrogen levels of approximately 60 - 80 per cent. Changes in the concentration form of nitrogen show that applied nitrogen was nitrified with substantial nitrogen removal due most to denitrification.

Phosphorus levels in the primary effluent averaged 11 mg/l, of which 80 per cent was orthophosphate. Groundwater total phosphorus levels ranged between 1.0-2.0 mg/l, slightly above the 0.7 mg/l background level. Orthophosphate was generally less than 0.4 mg/l in wells located around the treatment area but was higher than the 0.1 mg/l background level. Chloride, sulfate, pH, and hardness levels varied little from observed effluent levels, although conductivity and alkalinity were substantially less.

Analysis of groundwater samples for fecal coliform bacteria proved negative. Mean numbers of total coliform bacteria in the groundwater were less than 200/100 ml although two wells showed mean counts of 400-600 organisms per 100 ml of samples.

## INTRODUCTION

Treatment of municipal wastewater using the biological, chemical and physical process of the soil is an alternative to conventional secondary and tertiary wastewater treatment facilities. Spray irrigation, overland flow, and rapid infiltration has been used to treat wastewater in the past. Each approach has different design and operational criteria which depend on site characteristics and other local factors. Where agricultural and open lands are unavailable for spray irrigation or overland flow, rapid infiltration using naturally occurring sand and gravel deposits may provide a workable and efficient method for treating municipal and industrial wastewater.

With proper operation and maintenance, rapid infiltration beds have provided long-term service with substantial wastewater renovation (7, 8, 10, 11, 15, 23). Important to maintaining renovation capability is the operational cycle during which the treatment bed is alternately inundated and then permitted to rest. The inundation period varies depending on the geographical location, soil characteristics and climatic conditions. Regularly scheduled intermittent rest periods for the basins are necessary to permit recuperation of infiltration rates which degenerate over the inundation period. Physical and biological soil clogging results in a reduction of infiltration rate (3, 8).

Physical clogging results from the transport and deposition of particulate matter in the treatment bed. Lodging of suspended solids, usually of an organic nature, between soil particles, leads to partial or complete blockage of soil pores, thereby decreasing substantially the effective pore size, thus reducing the infiltration rate. Eventually, an organic layer forms on or near the surface horizon. Once formed, the organic layer reduces oxygen transfer into the soil and decreases the effectiveness of the soil system to renovate wastewater.

Biological clogging results also from microbial growth on the filtered organic matter and on soil particles. McGauhey and Krone (18) concluded that various micro-organisms and microbial by-products were responsible for the biological clogging of the soil system. Fortunately, during the rest period, biological and chemical processes re-establish most of the

soil's original infiltration capacity. Biological utilization of filtered organics is dependent upon the nature of the organic materials and oxygen availability. Where ample oxygen is present, the easily biodegradable substances, such as sugars or starch, are readily assimilated. Tannins, lignins and waxes are not easily degraded and tend to concentrate (15, 18).

Because reclaimed waters from land treatment systems often recharge groundwater aquifers or surface water bodies, the quality of renovated water is an important consideration. Lavery (15) found that rapid infiltration and natural soil adsorption of pretreated wastewaters can result in a high quality water suitable for groundwater recharge. Bouwer (7) observed that rapid infiltration of secondary treated wastewaters removed essentially all suspended solids, biochemical oxygen demand and fecal coliform bacteria. The concentration of phosphorus, about 13 ppm in the secondary effluent, was reduced to a range of 5-7 ppm after filtration. Similarly, Amramy (3) observed a significant reduction in BOD and phosphate levels while COD was reduced from 172 ppm to 85 ppm.

Nitrogen levels in applied effluents and subsequently in reclaimed water from land treatment system remain a major consideration where these waters augment public water supplies and where eutrophication of surface water is a problem. Nitrogen from primary sewage effluents is mostly in organic and ammonium nitrogen, while nitrogen in the many secondary effluents is predominantly in the ammonium form. Laboratory and field investigations under aerobic conditions have shown that organic and ammonium nitrogen applied to treatment beds is readily converted to nitrate (8, 12, 13). However, not all nitrogen applied finds its way into underlying groundwater. Bouwer (7) observed a 70 per cent reduction in reclaimed water in comparison to nitrogen in the applied secondary effluent. Mechanisms for possible nitrogen removal have been discussed by Bouwer (7) and Lance (12). Volatilization of ammonia from wastewater effluents could remove some nitrogen. Because ammonia volatilization is pH dependent, significant amounts of ammonia would not be volatilized unless the effluent pH was made more alkaline by lime additions. Ammonium adsorbed by the negative charged clay and colloid particles of soil could remove considerable amounts of ammonium, but the quantities of ammonia added to the rapid infiltration beds in the effluent would saturate the adsorption capacity in a few years.

Biological uptake and incorporation of nitrogen into cell tissue could account for substantial nitrogen removal. Unless the biological material is removed from the application area, the nitrogen would be released back to the system upon microbial decay of the cell tissue. Ammonium

fixation by organic complexes could remove large quantities of nitrogen, but the fixation mechanism has limited capacity which would be satisfied after a short time by the large nitrogen additions to the rapid infiltration system. Bionitrification in rapid infiltration systems is possibly the most important removal mechanism. Bouwer (7) observed that nitrogen removal by denitrification was distinctly possible in land treatment systems and where oxygen was limited and both nitrate and organic matter were present. By this mechanism, nitrate is reduced mainly to nitrogen gas which escapes into the atmosphere. Although the denitrification process has not yet been positively demonstrated in rapid infiltration systems, substantial nitrogen removal has been attributed to it (17, 11).

Field studies of rapid infiltration systems have been conducted on new facilities. These investigations demonstrated that the wastewater treatment technique can provide acceptable renovation on a short-term basis. However, questions pertaining to long-term treatment effectiveness of rapid infiltration systems for treating primary sewage effluents still remain.

#### STUDY OBJECTIVES

Investigations were begun in October 1972. The objective was to determine the treatment effectiveness of a rapid infiltration system to renovate unchlorinated primary sewage effluent after more than 30 years operation in a northern climate, and to ascertain any physical and chemical changes in the soil of the treatment beds.

The study was sponsored by Cold Regions Research and Engineering Laboratory as part of RDTE Protect 4A062112A891, Task 05, Environmental Quality for Army Construction. It was conducted in cooperation with the Massachusetts Agricultural Experiment Station and the Department of Plant and Soil Science, University of Massachusetts, Amherst.

#### DESCRIPTION AND OPERATION OF TREATMENT FACILITY

Ft. Devens is a U.S. Army Military Installation located about 52 km northwest of Boston and 35 km northeast of Worcester, Massachusetts. The Post includes portions of four Massachusetts communities: Ayer, Harvard, Lancaster and Shirley. The number of persons residing or working on the Post has fluctuated with the mission of this installation. In 1973, the day-time population was about 15,000 of which 10,400 were permanent residents.

The present sewage treatment facility was constructed in 1942 (Figure 1). During a two-year period, 1 September 1946 - 31 August 1948, when military activities were curtailed and the Post was used as the Fort Devens Extension of Massachusetts State College at Amherst (now University of Massachusetts), the sewage treatment facility continued to provide service. Sewage from the cantonment and housing area is carried in sewer lines by gravity flow except for several small pumping stations required to lift the wastewater at several points on the Post. By the time the sewage is transmitted to the main pumping station, kitchen grease and fats and various oils have been removed in grease traps. These traps are cleaned periodically and collected materials are deposited in sanitary landfills.

At the main pumping station, the sewage passes through a comminutor and is then pumped approximately 450 meters to three Imhoff tanks where the wastewater receives primary treatment. In the Imhoff tanks, gross solids settle from the influent. Retention time in the tanks is about six hours based on a design flow volume of 11,355 m<sup>3</sup>/d (3.0 MGD). However, retention varies in relation to influent flow volume and sludge accumulation.

Daily wastewater flows through the treatment facility between 1 January 1960 and 31 December 1973 have varied considerably (Figure 2). Approximately 68 per cent of the daily flows observed from 1 January 1968 through 31 December 1973 were within a range of 4,057 to 3,063 m<sup>3</sup>/day. Larger daily flows generally occurred from February to mid-April while smaller volumes were recorded during August through December (Appendix A, Figures 1 through 6).

Settleable materials or "sludge" accumulate on the bottom of the Imhoff tanks and are gravimetrically withdrawn to sludge dewatering beds during April and November of each year. These dewatering beds are equipped with underdrains which discharge collected water to the adjacent wetland area.

Unchlorinated effluent from the Imhoff tanks is discharged to 22 treatment beds, each about 0.32 hectare (ha) in size. The beds are constructed on a large oval-shaped steep-sided kame, composed of stratified sands and which rises approximately 21 meters above the flood plain of the Nashua River. Normally, the treatment beds are used in a rotation sequence which consists of three beds concurrently being flooded with effluent for a two-day application period which is followed in sequence by a 14-day "rest" or "dry-up" period. During holidays and weekends, anticipated effluent flows and precipitation may necessitate using four

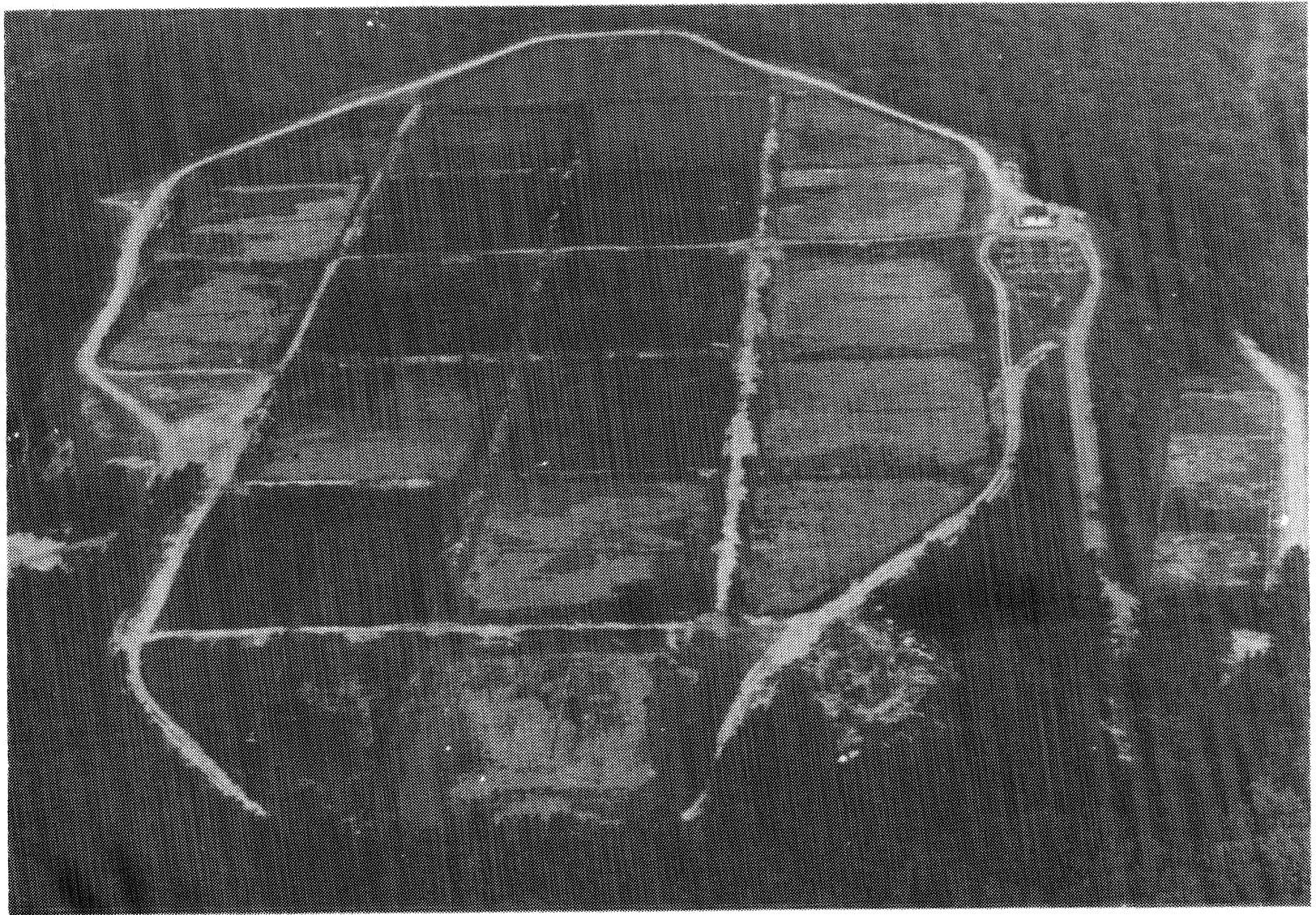


Figure 1. Aerial photograph of sewage treatment facility, Ft. Devens, Mass.

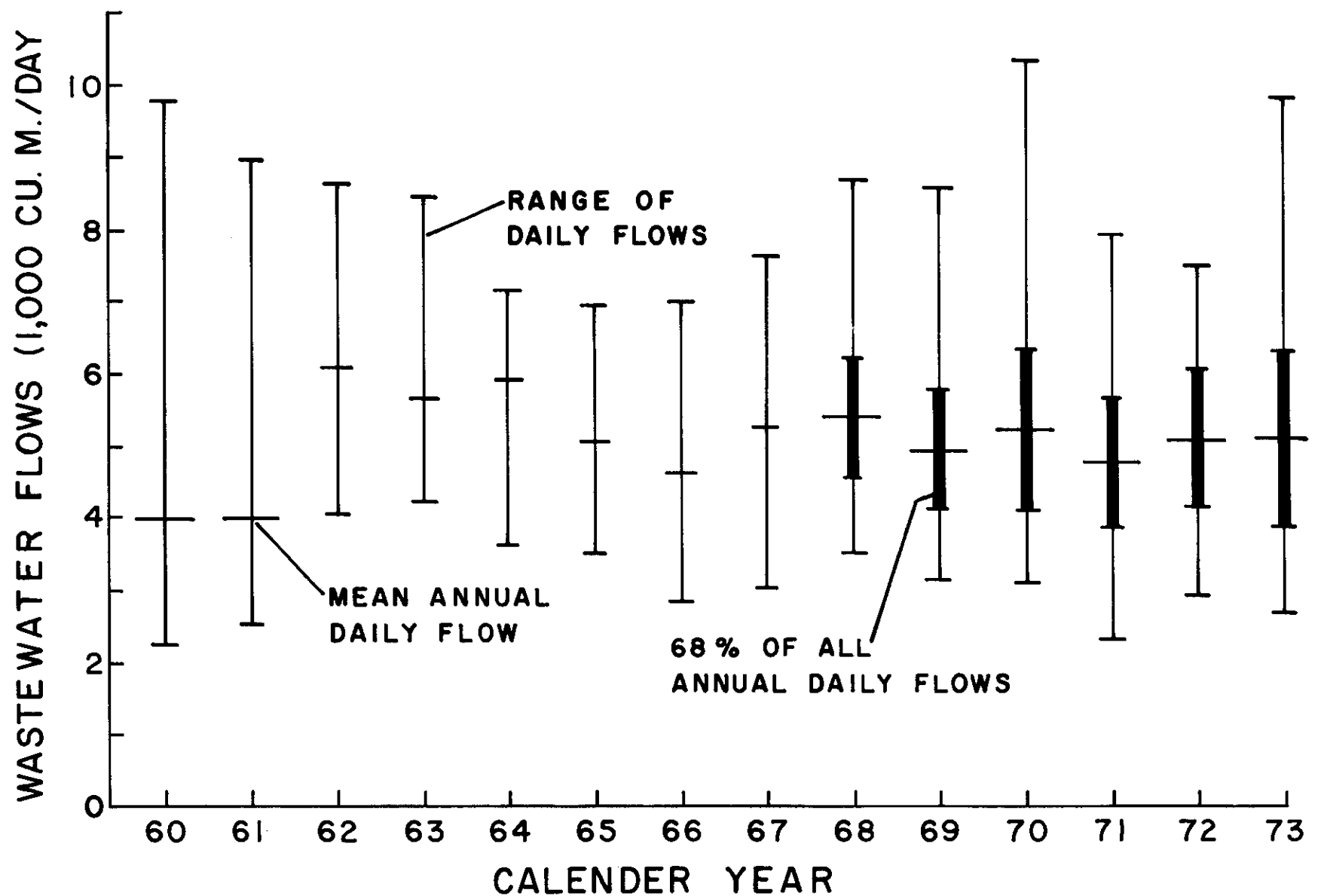


FIGURE 2: Mean annual daily wastewater flows and annual range of daily wastewater flows (1960-1973).

filter beds rather than the normal three (Figure 3).

Based upon the application cycle, each treatment bed receives primary sewage effluent approximately 52 days each year. Assuming a mean annual effluent flow of 5,061 m<sup>3</sup>/day (1.34 MGD) and equal effluent distribution per unit area, effluent additions to the beds have averaged about 28.3 meters per annum (93.8 ft/yr) since 1960.

During the application period, effluent may accumulate on the bed to a thickness of 15 - 50 centimeters (0.5 - 1.6 ft). All effluent infiltrates the bed within the initial two or three days of the "recovery" period, thus exposing the bed surface to the atmosphere.

Ice and snow often accumulate on the beds during the winter and can reduce infiltration somewhat. The effluent is sufficiently warm (8 - 12°C; 46 - 54°F) during this time that it can melt the ice and snow cover and move into and through the treatment beds.

During the summer, each treatment bed has a good stand of grass vegetation. The major portion of the biomass consists of annual "weedy" grasses; fall panicum (Panicum dicotomiflorum Michx.) and barnyard grass (Echinochloa crusgalli (L.) Beauv.). Other grass species, as well as Carex species, have been observed growing on the beds.

Maintenance procedures prescribed for the treatment beds required the surface of each treatment bed be excavated periodically to a depth of 0.3 meters (one foot) and this material be replaced with bank run sand and gravel. Prior to replacing the excavate, the exposed surface is harrowed or raked. The most recent "cleaning" and "grooming" operation was completed in October 1968, at which time eighteen of the twenty-two treatment beds were excavated to a depth of 0.45 to 1.22 meters (1.5 - 4.0 ft). Excavation below the specified 0.3 meter (one foot) depth was necessary to remove a "tarlike" layer about 0.45 meters (1.5 ft) in thickness which developed in the treatment beds.

The operation and maintenance of the treatment facility is carried out by two full-time employees. While the treatment facility including the effluent application area was designed to handle a given wastewater flow, the application of the daily flows to various combinations of treatment beds was developed from operation experience over an unknown period. The basis for the application cycle is the volume of primary effluent to be treated and the continued capacity of the beds to accept the effluent. Prior emphasis has not been towards the quality of the water resulting beneath the application area. As the quality of the water in the Nashua



River is improved, the quality of the percolate from the rapid infiltration beds will receive additional consideration.

## METHODS AND MATERIALS

The general approach taken in evaluating the impact of primary effluent to treatment beds and underlying groundwater was:

1. to compare physical and chemical characteristics of soil with material of similar texture and origin to which primary effluent had never been applied, and
2. determine concentrations of selected water quality parameters in regional groundwater, primary effluent, and groundwater beneath and surrounding the land treatment area over a one-year period.

### a. Soil Investigations

Continuous dry soil samples were taken from two 6.1 meter cores (20 ft) in four treatment beds, numbers 13, 14, 15 and 16 (Figure 3) for physical and chemical determinations. To obtain background characteristics, samples were taken from the upper 6.1 meters of an adjacent sand and gravel deposit. The glacial formation was similar in elevation and origin to the formation on which the treatment beds were constructed. Samples were also obtained from well #2 (Figure 3) to a depth of 24.4 m for physical determinations. Each boring was made using a pneumatic drill and a 6.3 cm (2.5 inch) diameter, 1.52 meter (5.0 ft) sampling spoon. Each core sample was subdivided into horizons differing in soil texture. Coring below a depth of 3.05 meters (10 ft) required casing the core hole to prevent sidewall collapse. Each hole was cased with 7.6 cm (3 in) diameter steel-well casing which was pneumatically installed to 3.1 and 4.6 meter depths (10 and 15 ft) after soil samples were obtained. Samples were placed into labeled plastic bags and transported to the laboratory where they were stored at -10°C.

Before physical and chemical analysis, each soil sample was thawed and air dried at 25°C. Particle size distribution was determined by dry sieving with a standard series of nested of 20 cm (8 in) sieves (6). A roto-tap shaker was used to shake each sample for five minutes.

Water retention values at -0.1, -0.3, -1.0 and -15.0 bar tensions were run in triplicate for samples obtained from two 3.05 meter (10 feet) borings and one 24.38 meter (80 foot) boring in the application

area using the pressure plate and pressure membrane methods (6). A portion of the air dried soil sample passing a 2 mm (0.072 inch) sieve was placed within the rubber retaining rings, saturated with tap water, and allowed to equilibrate under saturated conditions before being subjected to the appropriate pressure.

Saturated hydraulic conductivities were determined using commercially available permeameters (Soil Test model K-620). A constant head was employed. Using Darcy's equation  $V = Q/AT = Ki$ , hydraulic conductivities were computed for samples from the 3.05 and 24.28 meter borings, where  $V$  is the velocity of flow (cm/sec);  $Q$  is the quantity of water discharged (ml) in time,  $T$  (seconds);  $A$  is the cross-sectional area of the permeameter (cm<sup>2</sup>);  $K$  is the hydraulic conductivity in (cm/sec) and  $i$  is the hydraulic gradient (dimensionless).

Three different kinds of infiltration tests involving five treatment beds were conducted. Metal ring infiltrometers, 38 cm (15 inch) diameters were used at three sites in each treatment beds; nos. 13, 14, 19 and 20. Infiltration rates were determined by measuring the amount of water entering through the soil surface inside the infiltrometer as a function of time. A head of approximately 5 cm was maintained inside the ring throughout each test. Primary sewage effluent from the treatment plant was used as the infiltration water. Triplicate soil samples were taken at each site and tested for water content determination. After infiltration rate determinations were made on bed 14 using metal ring infiltrometer, bed 14 was subjected to a flooding test where all of the effluent flow was directed to this single bed for a seven-hour period. Infiltration rates was determined as a function of time by measuring the difference between effluent volume increase in the bed and total inflow to the bed. During the flooding test, water level readings were taken at five locations in the bed, every 30 - 40 minutes. Because inflow exceeded infiltration, the infiltration rates were influenced by a slow increase in pressured head.

The third method employed to determine infiltration was to observe effluent seepage in bed 1 at the completion of the inundation period. Readings of effluent level were taken at two locations at approximately 40 minute intervals.

Chemical analyses were conducted on samples obtained from two borings in filter beds 13, 14 and 15 which had been stored as described above. Each sample was thawed, air dried at 25°C and that portion passing through a 2 mm sieve was evaluated for: pH, 1:1 ratio of sample to water mixture using a Beckman Zermatic pH meter; organic matter by the ignition method (4); electrical conductivity, 1:2 ratio of

sample to water mixture (17); total nitrogen by the Kjeldahl method (4); phosphorus by the quinolinium molydophosphate method (4); orthophosphate by the citrate extraction (4); total amounts of calcium, magnesium, potassium, manganese, copper, iron and zinc by atomic absorption spectrophotometry on HCl acid extract (4); boron by the Quinalizarian method (4); sulfur by the potassium nitrate-nitric acid digestion method (5); chlorides by the mercuric nitrate method (4); and cation exchange capacity (4).

b. Groundwater and Effluent Investigations

Monitoring the quality of groundwater below the treatment beds and in the <sup>area</sup> peripheral to the application site was accomplished by collecting and analyzing liquid samples from fourteen groundwater observation wells. Six wells (nos. 3 - 8) consisting of 3.2 cm (1.25 in) diameter metal riser pipe and 0.91 m (3.0 ft), # 10 metal well screens were installed during October 1972. Six additional wells (nos. 9 - 14) consisting of PVC riser pipe and 3.05 m (10 ft), #20 PVC well screens were added during April 1973 (Figure 3). Each well screen was positioned so the upper portion of the water table could be sampled throughout the hydrologic year. Depth to water table in wells #1 and #2 was too great for suction pumping, therefore, 7.6 (3 in) and 15.2 cm (6 in) diameter metal riser pipe and 1.52 m (5 ft) well screens were installed.

All wells were installed by wash drilling, using either skid mounted or truck mounted diamond core drills. All holes were cased during the drilling operations with NW and SW size flush joint casings. Tap water was used in the installation of casing for removing substrate from the casing during the drilling operation in order to prevent possible contamination. Following installation, each well was pumped for 30 minutes to remove any fine sediments present in the well.

Groundwater samples were collected biweekly from each observation well. Sampling the larger diameter deep wells was accomplished using a 400 ml (0.11 gal) Kemmerer water sampler. Samples from the shallow 3.2 cm diameter wells were collected directly into sterile, 1000 ml (0.26 gal) labeled fleakers using a hand-operated suction pump.

A composite sample of primary effluent was collected during the 24-hour period preceding each biweekly well sampling. The composite sample was collected with a proportional sampler equipped with a refrigeration unit which cooled the effluent to 5°C. Following thorough mixing of eighteen liters of composited effluent, a 1000 ml (0.26 gal) subsample was drawn off for laboratory analysis.

Groundwater and composite effluent samples were transported to the laboratory within four hours after collection for immediate chemical and biological analyses. All samples were analyzed for the following parameters according to procedures outlined in Standard Methods (2): pH (glass electrode method); electrical conductivity (Wheatstone bridge); alkalinity (potentiometric); five-day biochemical oxygen demand (Winkler determination); organic Kjeldahl nitrogen (potentiometric titration); ammonia nitrogen (Nesslerization); nitrite nitrogen (colorimetric); total and ortho-phosphates (stannous chloride method); sulfate (turbidimetric); and total coliform bacteria (membrane filter technique). Hardness (20), nitrate nitrogen (14), and chlorides (21) were all determined utilizing specific ion electrodes.

## RESULTS AND DISCUSSION

### a. Soil Investigations

Physical Determinations - A stratigraphic section (Figure 4) constructed from the drillers' boring logs, depicts the various horizons in the upper 6.1 m (20 ft) of the filter beds (A-A<sup>1</sup>). The formation in which the treatment beds were constructed is primarily poorly graded sands or gravelly sands, with interspersed lenses of silty sand and sandy gravels. The low-lying area adjacent to the beds consist of medium to fine-textured sands with some lenses of gravel.

Sieve analysis of the substrate samples from the treatment bed formation reveals that most strata contain only a small percentage (1 - 3 per cent) of particles smaller than 0.053 mm. Only seven of the 39 treatment bed strata samples contained more than five per cent "fines" (silt and clay). The tabular data (Appendix A, Table 1) suggest appreciable differences in distribution of particle size between the various horizons in the beds. About one-half of the strata sampled in beds 14, 15 and 16 were composed of over 40 per cent of particles falling into the gravel-size range (2 mm). Percentage of particles greater than 2 mm was inadvertently not determined on samples from bed 13.

Background samples were collected from a formation consisting of stratified horizons of sand and gravels. Coarse to medium sand comprises the major portion of the 3.05 meter profile (10 ft), with gravel forming an important percentage in the 0.3 - 0.9 (1 - 3 ft) meter horizon. Below the 0.9 meter depth, the formation consisted of two meters (6 ft) of medium sand which was underlain by strata of sandy gravel and gravelly sand, similar to the material encountered in the well #2 profile. Silts

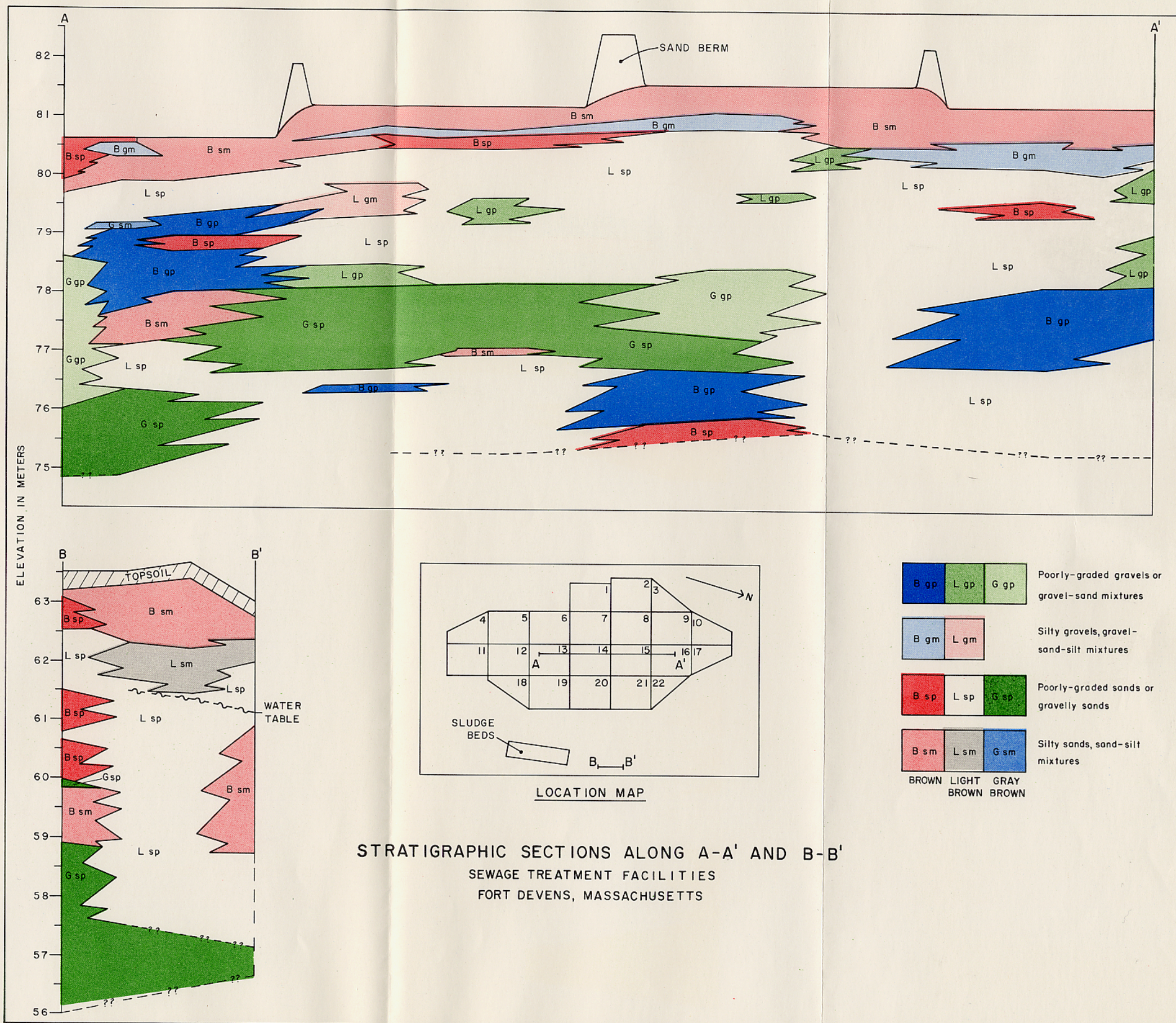


FIGURE 4:

and clays encompassed about 10 - 15 per cent of the volume in the upper profile but less than one per cent below the 0.5 meter (1.6 ft) depth.

Water retained by boring samples from beds 14 and 15 and well #2 were similar and representative of unconsolidated coarse textured material with low percentages of silt and clay sized particles (Figure 5). The water retained in the samples decreased rapidly from saturation (about 55 per cent water) to -1.0 bar tension (generally 1 - 2 per cent water). Moisture content changed very little between -1.0 bar and -15.0 bar tensions. Water retained in each sample at the -15 bar was less than 1.0 per cent. Moisture retention was influenced more by the percentages of silt and clay than the larger grain-sized particles. Samples with four per cent fines held more moisture at the -0.1 and -0.3 bar tension than samples with lesser amounts of silt and clay.

During an effluent flooding cycle, the surface soil approaches saturation and then drains to field capacity during the recovery period. Assuming a bulk density of  $1.2 \text{ gm/cm}^3$  and that a tension of -0.3 bars approximates field capacity, then the average depth of water held in various horizons would be about 0.5 mm/cm soil depth.

This means that following the rest period, there is sufficient void space in about 3m (10 ft) of soil to hold all effluent which would be applied during a three-day application period to three treatment beds. Since water held in the soil at tensions less than about -0.3 bars will move through the sand and gravel horizons, the treatment beds are "dewatered" during the recovery period as water percolates through the formation eventually adding to groundwater.

Hydraulic conductivities of samples from well #2 and treatment bed 15, which had passed through the 2 mm sieve, were measured using a saturated column under a constant head. Permeabilities were of the order  $10^{-3}$  to  $10^{-2} \text{ cm/sec}$  ( $4 \times 10^{-4}$  to  $4 \times 10^{-3} \text{ in/sec}$ ) (Table 1). Most permeabilities could be classified as rapid to very rapid with only a few moderately permeable horizons. Moderately permeable strata were generally those with greater than four per cent "fines" and 10 per cent very fine sand. Rapid or very rapid permeabilities were generally observed in material with less than 10 per cent particles smaller than 0.10 mm. Theoretically, each horizon will easily transmit the designed daily wastewater flow of  $11,355 \text{ m}^3/\text{d}$  (3 MGD) provided a one hectare (2.47 acre) application area is used. The occurrence of effluent ponding on the treatment bed surfaces show in fact that the infiltration rate decrease during application period, thus limiting the rate at which effluent moves into and through the formation.

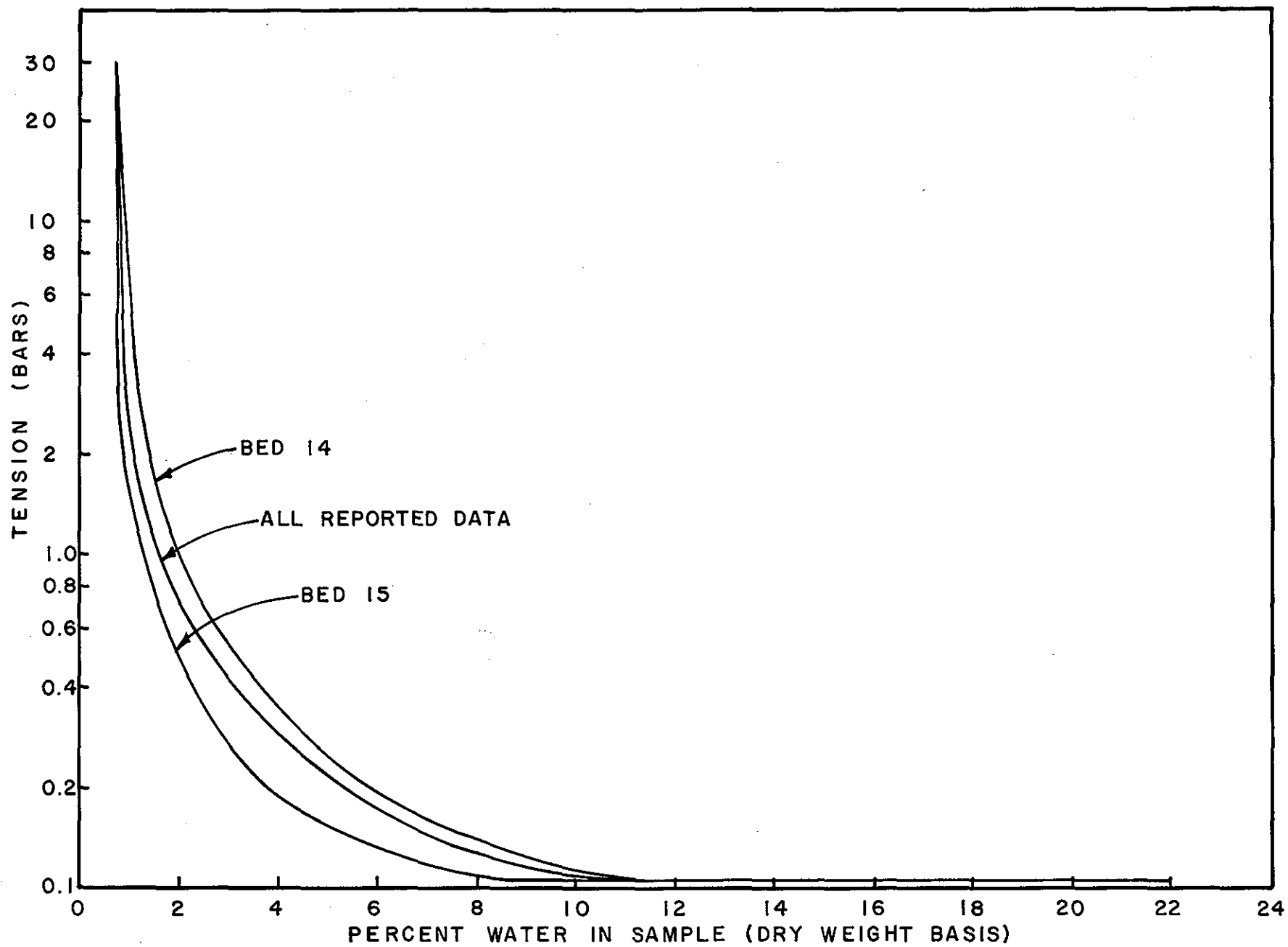


FIGURE 5: Moisture release curves of substrate samples.

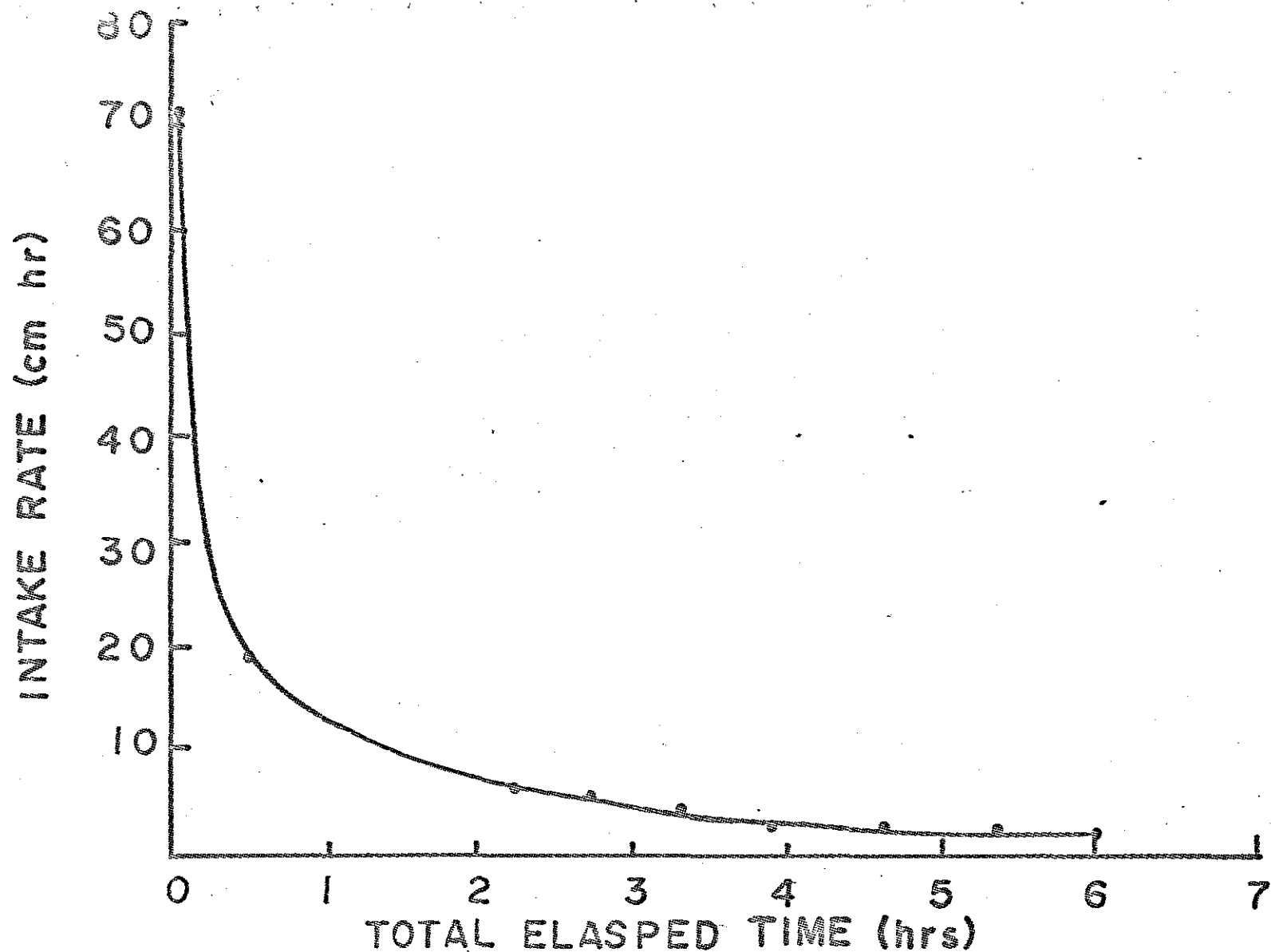


FIGURE 6 INTAKE RATE OF TREATMENT  
BED 14 AS A FUNCTION OF TIME

TABLE 1

HYDRAULIC CONDUCTIVITIES  
OF  
SUBSTRATE SAMPLES FROM INFILTRATION AREA

Sample Depth  (meters)	Hydraulic Conductivity (cm/sec) $\times 10^{-2}$
<u>Filter Bed 15</u>	
0.0 -0.15	1.14
0.15-0.46	0.80
0.46-1.22	1.75
1.22-1.52	1.90
1.52-3.05	0.14
3.05-4.27	0.46
4.27-4.57	0.42
4.57-4.88	1.78
4.88-5.79	0.15
5.79-6.10	0.20
<u>Well #2</u>	
0.0 -0.24	0.28
0.24-1.28	0.19
1.28-1.52	0.20
1.52-2.71	0.80
2.71-3.05	0.17
3.05-4.57	0.14
6.10-7.62	0.83
7.62-9.14	0.03
9.14-10.67	1.70
10.67-12.19	0.39
12.19-13.72	1.00
13.72-15.24	0.60
15.24-16.74	0.70
16.74-18.29	0.21
18.29-19.81	0.23
19.81-21.34	2.40
22.86-24.38	0.10

Infiltration tests were conducted on five treatment beds (beds 1, 13, 14, 19 and 20) which represented a wide variety of soil moisture conditions and bed surface conditions. Soil moisture ranged from near saturation to very low water contents and surface conditions consisted of an organic mat overlying the soil surface or an exposed bare soil. At the time infiltration tests were performed, the following conditions existed; the two-day flooding cycle had just been completed for bed 1 and it was under about 50 cm of effluent; the 14-day rest period of beds 13 and 14 had just been completed, representing minimum moisture in the surface soil; and beds 19 and 20 were about 30 hours into their rest period, representing a condition where surface soil of these beds was free from standing water but still quite wet compared to beds 13 and 14. All beds except bed 19 had been "cleaned" in 1968.

Because effluent intake rate depends on moisture content, measurement of water content of the surface soil were made at each site where infiltrometers were used. Data in table 2 summarizes the results of the metal ring and infiltrometer tests in beds 13, 14, 19 and 20. These data show the effect of initial soil moisture content and time on effluent intake rate. Treatment beds with slowest infiltration rates, where those having highest initial soil moisture content levels.

Measured soil moisture contents of the surface 0.0-0.15 m (0-6 in) in beds 13 and 14 are greater than water retention data would suggest, probably because of high organic matter incorporated in these soil samples. Bed 13, having the lowest initial water content, also had the greatest effluent intake rate.

Data in Table 3 and in Figure 6 show the effect of time on effluent intake rate in bed 14. Because all effluent flow during a seven-hour period was directed to bed 14, intake rates do not represent rates under actual management practices where effluent is simultaneously applied on three beds and pressure head build-up is slower. Under normal operating conditions, it take longer than six hours for the intake rate to approach 2.2 cm/hr or the time when ponding starts.

Ponding on treatment beds begin when effluent flow from the treatment plant exceeds intake rate of the treatment beds. Seepage test conducted in bed 1 suggests that at completion of the inundation cycle following actual management practices, effluent intake rate has decreased to about 1.5 cm/hr or lower (Table 4). It should be remembered that during both flooding and seepage tests and during normal flooding of the beds, that the intake rate is influenced (increased) due to increased pressure head that develops as effluent ponds on the treatment bed.

Table 2.

Initial (0-3 minutes) and average (3-60 minutes) effluent intake rates (I.R.) and water contents (dry weight basis) of four treatment beds.

<u>Bed</u>	<u>Site 1</u>	<u>Site 2</u>	<u>Site 3</u>	<u>Average</u>
<u>Bed 13</u>				
% H <sub>2</sub> O	2.4	3.8	3.1	3.1
0-3 min I.R. (cm/hr)	143	117	143	134
3-60 min I.R. (cm/hr)	49	35	33	39
<u>Bed 14</u>				
% H <sub>2</sub> O	6.2	6.7	14.5	9.1
0-3 min I.R. (cm/hr)	107	97	10.2	71.4
3-60 min I.R. (cm/hr)	12.2	40.4	2.6	18.4
<u>Bed 19</u>				
% H <sub>2</sub> O	18.0	9.7	15.5	14.4
0-3 min I.R. (cm/hr)	15.3	15.3	15.3	15.3
3-60 min I.R. (cm/hr)	3.8	4.9	3.4	4.0
<u>Bed 20</u>				
% H <sub>2</sub> O	12.0	12.7	12.8	12.5
0-3 min I.R. (cm/hr)	12.7	14.2	5.3	10.7
3-60 min I.R. (cm/hr)	5.7	4.2	6.0	5.3

Table 3

Effluent intake rate of treatment bed 14 as determined by flooding test a/.

Elapsed Time (hrs)	Intake Rate (cm/hr)
2.2	6.0
2.8	5.8
3.3	4.3
3.9	2.5
4.6	2.5
5.3	2.4
6.0	2.1

a/ Two hours were required to completely inundate the soil surface of treatment bed 14.

Table 4

Effluent intake rate to treatment bed 1 following two days inundation a/.

<u>Time</u> <u>(hrs)</u>	<u>Intake Rate</u> <u>(cm/hr)</u>
0.8	1.53
1.8	1.47
2.9	1.37
Average	1.46

a/ Intake rates are average values for the corresponding elapsed time in since measurements were started.

Data in Table 4 suggests that the rate intake could decrease to about 1 cm/hr or lower at the end of a flooding cycle when the pressure head has diminished to near zero.

Several factors are responsible for decreasing intake rates with time. Most obvious reason for the decline is because soils become water filled which causes increased frictional resistance to water flow. Some reduction in intake rate with time is no doubt due to physical and biological clogging.

Visual observation or approximately 48 corings made to depths of 0.3, 3.1 and 6.2 m (1, 10 and 20 ft) in treatment beds 13, 14, 15 and 16 failed to indicate the formation of a horizon, capable of limiting vertical movement of water. It should be pointed out that observations made in treatment bed 19, one of four beds not excavated in 1968, revealed the presence of a very definite "black tarlike" horizon which had a tainted sulfide and petroleum odor. This horizon was approximately 0.45 m (1.5 ft) thick and had formed at a depth of about 0.45 to 0.60 meters (1.5 - 2.0 ft). Formation of the horizon most probably required many years. Prior to 1968, the periodic "grooming" of the beds included the removal and replacement of the upper 0.3 m (1 ft) of sand and gravel material. This practice would have permitted the horizon to form. During this same time, there were numerous reports of crank case oil and kitchen grease flowing through the Imhoff tanks to the treatment beds. In 1968, although the two days inundation followed by 14-days recovery, application cycle was being practiced, the movement of effluent into the beds was reportedly negligible. All beds had standing water on them at the conclusion of the recovery period. The "black layer" was reportedly the primary factor for the standing water.

Following the excavation of the treatment to a depth below this horizon, the infiltration rate was restored. Infiltration of the effluent has apparently recovered in those beds not excavated in 1968. The black horizon at present does not form a continuous strata beneath the entire infiltration area in these beds. It is possible that the "black layer" has been decaying since 1968, or the reduced infiltration rate was related to other factors stemming from the continuous inundation of the treatment bed. In any event, correcting the dumping of oil and grease into sewer lines plus better care of grease traps, has undoubtedly prevented the flow of oil and grease to the treatment beds since 1968.

Chemical Determinations - Data from the chemical analyses of the treatment bed and background samples are tabulated in Table 5. An approximation of chemical characteristics of various horizons in the

**TABLE 5**  
**CHEMICAL CHARACTERISTICS OF SOIL SAMPLES**

Sample Depth (meters)	pH	O.M. (%)	E.C. (mmhos/cm)	N (ppm)	P (ppm)	Ca (ppm)	Mg (ppm)	K (ppm)	Mn (ppm)	Na (ppm)	Cu (ppm)	Fe (ppm)	Zn (ppm)	B (ppm)	S (ppm)	Cl (ppm)	C.E.C. (meq/100g)
<u>Treatment Basin 13 a/</u>																	
0.0-0.15	5.7	1.23	0.23	460	472	488	712	365	36	21	29	3470	31	0.8	2.7	15	1.5
0.15-0.46	5.7	0.79	0.18	267	472	378	923	357	42	23	29	4143	26	0.6	1.8	14	1.0
0.46-0.61	5.8	0.76	0.18	252	550	350	1008	405	54	24	32	4340	25	0.8	2.0	13	1.0
0.61-0.91	5.8	0.61	0.15	150	397	295	1075	405	52	22	16	4480	22	0.8	2.0	12	0.8
0.91-1.52	5.8	0.75	0.16	132	541	591	2278	772	78	27	46	7045	41	0.8	3.0	13	1.3
1.52-1.98	6.3	0.54	0.08	113	359	467	1227	590	55	23	22	4813	25	0.8	2.3	13	0.8
1.98-2.59	6.2	0.59	0.12	92	414	582	2018	835	82	26	23	6202	37	0.8	2.7	11	1.1
2.59-3.05	6.2	0.76	0.12	105	552	958	2640	1520	107	26	25	11375	39	0.9	3.0	11	1.3
<u>Treatment Basin 14 a/</u>																	
0.0-0.30	5.6	1.13	0.17	417	571	1020	863	703	83	23	18	8385	37	3.4	2.0	5	1.4
0.30-0.61	5.8	1.00	0.15	382	623	722	625	625	78	25	14	6750	32	2.5	2.1	8	1.4
0.61-1.22	6.0	0.88	0.11	215	599	600	664	847	94	21	12	7124	30	2.4	2.0	5	1.2
1.22-1.52	6.2	0.90	0.08	684	537	726	1133	1100	106	23	16	8250	43	2.0	1.6	6	1.5
1.52-2.74	6.3	0.62	0.06	131	434	706	830	1082	86	25	14	6858	62	2.0	2.0	6	1.0
2.74-3.05	6.4	0.72	0.07	126	529	1369	1440	1652	1162	25	18	11188	42	1.8	2.0	5	1.2
<u>Treatment Basin 15 a/</u>																	
0.0-0.15	5.6	1.39	0.21	531	691	970	2002	528	73	22	19	7435	43	1.1	3.1	13	1.4
0.15-0.46	6.0	1.08	0.13	283	796	819	1594	695	84	14	14	3750	37	0.9	3.1	7	1.4
0.46-1.22	5.9	0.93	0.16	231	669	552	1524	806	80	16	11	6029	28	0.8	3.1	6	1.3
1.22-1.52	6.1	0.71	0.10	112	412	482	1688	834	83	19	9	6158	23	0.3	3.3	6	1.1
1.52-3.05	6.2	0.55	0.09	82	368	480	1516	784	80	10	17	6188	22	0.4	3.3	5	0.8
<u>Background Samples</u>																	
0.0-0.15	4.4	5.41	0.03	920	299	108	484	188	79	5.3	4.0	8469	13	4.5	1.3	10	8.8
0.15-0.30	4.6	2.36	0.0	270	301	66	781	200	60	4.5	4.0	10438	12	3.8	1.2	6	2.4
0.30-0.46	4.6	1.77	0.0	210	303	60	1229	275	73	4.0	6.0	9188	13	5.3	0.6	4	2.3
0.46-0.61	5.0	0.45	0.0	80	153	59	760	344	69	3.3	4.0	3688	8	4.1	0.4	4	1.0
0.61-0.91	4.8	0.43	0.0	60	160	78	693	300	48	3.3	4.0	3500	7	4.1	1.5	4	0.9
0.91-1.83	5.2	0.26	0.0	35	146	132	718	425	66	3.2	4.5	3094	8	0.8	0.6	2	0.6
1.83-2.44	5.2	0.39	0.0	50	228	186	896	437	75	2.9	5.0	4500	9	1.9	0.4	2	0.8
2.44-3.05	5.2	0.40	0.0	60	304	321	729	519	62	2.4	3.5	4406	10	6.2	0.6	2	0.9

a/ Average of data from two cores.

treatment beds before primary effluent was applied either after construction or the cleaning and grooming operation are best shown by comparing data for background samples collected at depths greater below 0.6 meters.

From this portion of the background formation, bank run of sand and gravel for replacing the excavate removed during the "cleaning" operation was obtained. Chemical analysis of the soil samples collected in the treatment beds show the beds are acidic in nature, pH 5.6-6.4 throughout the 3.05 meter (10 ft) profile. Compared with the acidity of background samples (pH 4.4-5.2), the treatment beds are less acidic. Organic matter in the treatment beds was substantially higher than the 0.4 per cent observed in background soil samples. Surface horizons in the treatment beds ranged from 1.2-1.4 per cent. Organic matter levels decreased to about 0.7 per cent at the 3 meter depth (1 ft). Organic matter accumulation within the upper 0.15 meters (0.5 ft) of the treatment beds profile has been about 0.2 per cent per annum since the treatment beds were "cleaned and groomed" in 1968.

Conductivity of treatment bed samples ranged from 0.10 to 0.23 umhos with the highest values observed in samples from the surface horizons. Conductivity of background samples were less than detection limit below the 0.15 m depth (0.5 ft).

Cation exchange capacities of background soil samples varied from 2.0 to 8.0 meq per 100 grams in the surface horizons and about 0.5 meq per 100 grams at depths greater than 0.5 meters (1.6 ft). The higher exchange capacities in the surface horizons of the background formation appear directly related to the level of organic matter and the greater percentage of silt and clay size particles. Cation exchange capacities of soil samples taken in treatment beds were relatively constant throughout the three meter sampling depth and ranged from 0.79 - 1.51 meq per 100 grams. Compared to background samples from lower strata, the exchange capacity in the treatment beds has increased. It appears the exchange capacity in the treatment beds has increased during the six years since the beds were last renovated. It is uncertain at this time what factors have caused the increase but accumulation of matter at levels above those observed for background samples is probably an important factor.

Phosphorus concentrations in soils from treatment bed profiles range from 5570 - 7060 kg/ha-m (1515 - 1920 lb/ac-ft) which represent an increase of 250 per cent over observed background levels. Phosphorus levels in the surface horizons of treatment beds was slightly greater than levels in lower horizons. Even so, phosphorus levels were somewhat constant in the 3 meter (10 ft) bed profile. Approximately 60 per cent of

the total phosphorus observed in the treatment beds was orthophosphate as compared to 10 - 20 per cent in background samples. The form of remained phosphorus was not ascertained but is believed to be primarily phosphate precipitates of calcium, magnesium and iron. In bed 13, there appears to be a direct relationship between phosphorus concentrations and the concentration of calcium, magnesium and iron. This was not as apparent for other treatment beds.

Concentrations of calcium, magnesium, potassium and iron varied appreciably between different horizons and between the different beds. In general, these cations increase in concentration with increasing depth although some variations were observed. Compared to background sample levels of calcium, magnesium, potassium and iron in treatment beds were 300-500 per cent; 100-270 percent; 200-270 per cent; and 140-175 per cent greater respectively.

Sodium, copper and zinc concentrations varied between the treatment beds but were generally more uniform within the profile of each bed. Sodium ranged from 10-22 ppm; zinc from 22-62 ppm; and copper from 9-46 ppm. Compared to concentrations of these elements in background samples, treatment bed samples contained three to six times higher concentrations.

Chloride (5-15 ppm) and sulfur (1.6-3.3 ppm) levels in the treatment beds were about 200 per cent and 400 per cent respectively, greater than background levels. Concentrations of chloride and sulfur did not vary appreciably, however, within the treatment bed profiles. Chloride levels in bed 13 average about 8 ppm higher than background levels while chlorides in beds 14 and 15 were 1 - 3 ppm higher. Sulfur levels were 1-2ppm higher than background levels.

Boron concentrations ranged from 0.3-3.4 ppm in the treatment bed samples, however, these concentrations were about one-third the concentrations observed for background sand and <sup>gravel</sup> samples. Reduction in boron levels is due to the solubility of this ion which would have permitted it to be carried through the treatment beds in the percolating water (7).

Analysis of samples taken from the treatment beds showed total nitrogen was highest, 750-950 kg/ha in the surface horizon of each bed. Nitrogen levels decreased with increasing depth in treatment beds to approximately 180 kg/ha at 3.05 meter depth. Compared to nitrogen concentration in background samples, 72-90 kg/ha, total nitrogen in the surface horizons have increased eight to ten fold during the five-year

period since the beds were last excavated. Calculations show nitrogen accumulation has averaged about 680 kg/ha/yr (607 lb/ac/yr) in the surface meter of soil since the 1968 "cleaning and grooming" operation.

Table 6 summarizes the chemical changes in the rapid infiltration beds after receiving primary sewage effluent for thirty years. In comparison to the background soil samples, the treatment beds have substantially higher levels of most chemicals. Surface horizons of the treatment beds which had received primary effluent for only five years, had significant levels of organic matter, nitrogen, phosphorus, and other cations. Only boron appeared to have decreased from observed background levels.

#### b. Effluent and Groundwater Investigations

The volume of primary sewage effluent applied to the treatment beds in 1973 varied between 2676 and 9841 m<sup>3</sup>/day (0.7 - 2.6 MGD) with an average daily volume of 5049 m<sup>3</sup>/day (1.3 MGD) (Appendix A, Figure 6). Following the 16-day operation cycle, total volume of effluent applied during 1973 was almost 27.1 m.

Precipitation recorded at the treatment site during 1973 was about 112 cm (44 inches). Potential evaporation was estimated at about 66 cm per year (26 inches) which was apportioned monthly by the ratio of monthly pan evaporation to total annual pan evaporation (Table 7) (26). While this method may not portray actual water loss from the treatment site, it provides a reasonable estimate. Summing monthly precipitation and potential evaporation showed net monthly water loss from the site during June through September periods was 1.0 - 5.7 cm/mo while gains during October through May ranged from 1.6 - 9.6 cm/mo. Water gains and losses resulting from precipitation and evaporation were generally 1 - 2 per cent of the total water additions to the treatment site; however, above normal precipitation during December accounted for about 11 per cent of monthly total water.

Elevation of water tables in fourteen observation wells varied during the hydrologic year. Water tables observed in wells 3, 8 and 12 illustrates annual cycle observed in most wells (Figure 7). Water tables were closest to the soil surface during early spring but decreased to a minimum elevation during late fall.

The regional groundwater flow was generally towards the Nashua River and adjacent surface water bodies. Groundwater gradient between well 2 and the river was generally 1.0 - 1.6 per cent while gradients from the other wells to the river or adjacent surface water was about one

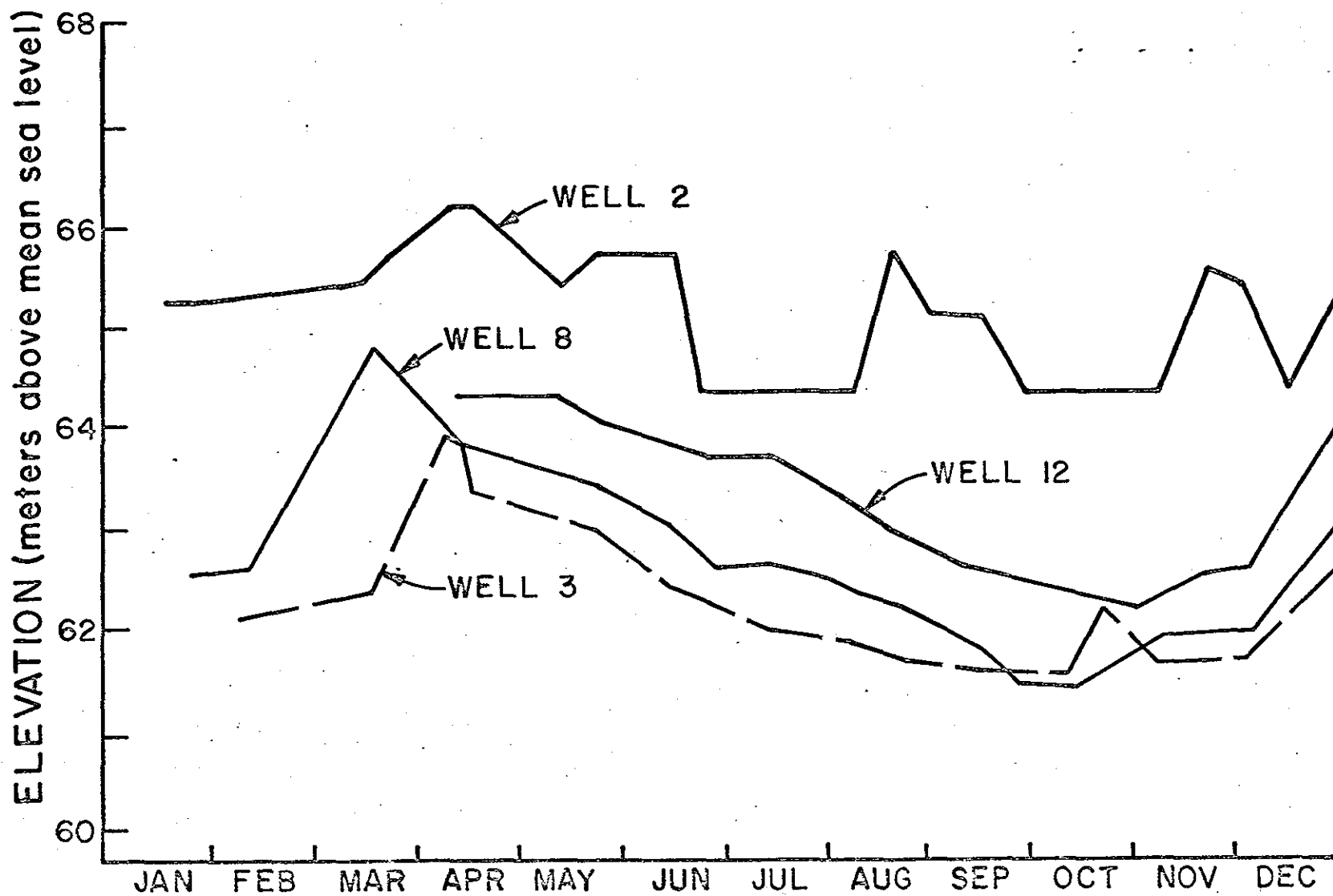


FIGURE 7: Water Table Elevation in Selected Wells during 1973

TABLE 6

SUMMARY OF SOIL CHEMICAL CHANGES IN THE  
TREATMENT BEDS COMPARED WITH BACKGROUND SOILS

Character	Change
pH	less acidic
organic matter	increased
electrical conductivity	higher
total nitrogen	higher
total phosphorus	+250%
calcium	+300-500%
magnesium	+100-270%
potassium	+200-270%
iron	+140-175%
sodium	higher
copper, zinc	higher
boron	decrease
chloride, sulfur	higher
cation exchange capacity	slight increase

Table 7

Water Additions and Losses to Treatment Beds (1973)

Month	Precipitation (cm)	Potential Evaporation (cm)	Effluent Additions (cm)	Total Water Inputs (cm)	Total Water Inputs x 100% Effluent Additions (per cent)
January	10.9	1.8	260	269.1	103.5
February	7.2	1.9	278	283.3	101.9
March	7.5	3.1	261	265.4	101.7
April	15.0	5.4	302	311.6	103.2
May	10.6	8.6	266	268.0	100.8
June	8.1	9.4	242	240.7	99.5
July	4.3	10.1	221	215.2	97.6
August	6.7	9.0	196	193.7	98.8
September	5.0	6.0	177	176	99.4
October	8.2	3.8	161	165.4	102.7
November	5.7	4.1	158	159.6	101.0
December	<u>22.4</u>	<u>2.8</u>	<u>185</u>	<u>204.6</u>	<u>110.6</u>
TOTAL	111.6	66.0	2,707	2,752.6	$\bar{X} = +1.7\%$

a/ Potential evaporation estimated at 66 cm/yr (26)

per cent. Water tables appeared to vary directly with water level in the river, which was approximately by the water table of well 3, about 74 m distance from the river.

Well 1, located west of the treatment site, contained measurable water table only from early spring to mid-summer. The water table elevation was 65 - 55 m (above mean sea level) between April and early July, before becoming dry.

The water table in well 2 remained relatively high throughout the study year, fluctuating between 64.5 - 65.6 m. While water table elevation may have been influenced by the river flow stage, it apparently was more affected by which treatment beds were inundated at a particular time. When beds in close proximity of well 2 were used, the water table was high, but decreased in elevation when beds more removed were utilized.

Chemical and bacteriological analyses of Imhoff tank effluent plus annual loadings of wastewater constituents to filter beds were summarized in Table 8. pH of primary sewage effluent ranged from 6.2-9.0 of type S lime to Imhoff tanks, primarily to maintain sludge pH may have helped maintain effluent pH within this range. Groundwater pH ranged from 6.1-6.7 (Table 9).

Wells 1 and 13 showed chloride levels ranged 8-20 mg/l which were representative of regional native groundwater. Motts and Saines (19) reported chlorides in groundwater aquifer used for municipal and domestic water supplies here after called native groundwater generally range 10-15 mg/l with some instances of 30-40 mg/l. Chloride levels in well 12 located 75 m from the treatment site ranged from 12-25 mg/l throughout most of the hydrologic year. However, beginning mid-November through December, chlorides concentration in well 12 increased to 120 mg/l. This increase was associated with a decline of water table elevation. Increased chloride levels reflect the influence of percolate from the treatment beds upon water quality and the direction of groundwater flow from the site. Although the water table in well 13 declined during this same period, chloride levels did not subsequently increase as observed in well 12. Chlorides in well 9 and 14 were a similar level, ranged from 70-300 mg/l with mean values of 144 and 162 mg/l, respectively. Chlorides in wells 3, 7, 8, 10 and 11 were the highest levels observed, 170-410 mg/l with mean values of 220-280 mg/l.

Although annual chloride additions to the treatment beds was about 41,000 kg/ha-yr chloride levels in the soil sample taken from

TABLE 8

CHEMICAL & BACTERIOLOGICAL CHARACTERISTICS  
OF IMHOFF TANK EFFLUENT AND  
ANNUAL ADDITIONS TO TREATMENT BEDS  
(1973)

Parameter	Effluent <sup>a/</sup>		Annual Loadings <sup>b/</sup> kg/ha <sub>c/</sub>
	Range	Mean	
pH (standard units)	6.2 - 8.0	-	-
Conductivity (umhos)	402 - 700	511	-
Alkalinity (ppm CaCO <sub>3</sub> )	116 - 245	155	42,560
Hardness (ppm CaCO <sub>3</sub> )	22 - 60	41	12,260
BOD <sub>5</sub>	30 - 185	112	30,750
COD	110 - 450	192	52,720
Total-Nitrogen	19 - 78	47	12,910
Organic-Nitrogen	11.5 - 32.8	23.4	6,425
NH <sub>4</sub> -N	6.2 - 42	21.4	5,880
NO <sub>3</sub> -N	0.4 - 2.8	1.3	360
NO <sub>2</sub> -N	0.002 - 0.06	0.02	5.0
Total PO <sub>4</sub> -P	6 - 16	11	3,020
Ortho PO <sub>4</sub> -P	3 - 15	9	2,470
Chloride	75 - 210	150	41,190
Sulfate	27 - 72	42	11,530
Total Coliform Bacteria x 10 <sup>6</sup> /100 ml	18 - 53	32	-

<sup>a/</sup> mg/l unless otherwise indicated

<sup>b/</sup> based on 5049 m<sup>3</sup>/day effluent flow volume

<sup>c/</sup> kg/ha x 0.8929 = pounds/acre

TABLE 9

CHEMICAL AND BACTERIOLOGICAL CHARACTERISTICS OF GROUNDWATER  
IN SELECTED OBSERVATION WELLS (AVERAGE VALUES)<sup>a/</sup>

Parameter	Well										
	1	2	3	7	8	9	10	11	12	13	14
pH (standard units)	7.3	6.8	6.3	6.4	6.6	6.5	6.1	6.3	6.6	6.2	6.5
Conductivity (umhos)	133	371	360	405	443	310	333	305	71	36	327
Alkalinity (mg/l CaCO <sub>3</sub> )	29	120	28	37	58	67	14	53	29	17	49
Hardness (mg/l CaCO <sub>3</sub> )	12	23	44	71	44	32	30	31	17	6	30
BOD <sub>5</sub>	3.5	12	2.5	2.1	2.0	1.4	0.9	0.8	1.2	1.0	0.9
COD	42	26	19	22	15	8	10	9	10	13	11
Total Nitrogen	1.3	14.5	19.5	28.0	19.8	10.4	20.3	12.1	3.7	1.9	9.7
Organic Nitrogen	0.5	8.3	2.3	3.4	4.2	3.7	1.2	1.0	0.8	1.2	1.5
NH <sub>4</sub> -N	0.6	5.3	1.3	4.7	4.5	3.2	0.5	1.0	0.3	0.3	0.4
NO <sub>3</sub> -N	0.2	0.9	15.6	19.5	10.7	3.5	18.6	10.1	2.6	0.4	7.8
NO <sub>2</sub> -N	0.01	0.03	0.3	0.4	0.4	0.02	0.02	0.02	0.01	0.01	0.02
Total PO <sub>4</sub> -P	0.4	5.9	0.9	0.8	0.8	1.4	1.3	1.9	0.6	0.7	1.1
Ortho PO <sub>4</sub> -P	0.1	5.6	0.2	0.1	0.1	0.4	0.1	0.2	0.1	0.1	0.1
Chloride	20	85	230	257	220	144	257	221	40	15	162
Sulfate	9	48	39	46	36	33	35	44	9	7	46
Total Coliform Bacteria (#/100 ml)	335	3900	210	110	158	230	620	130	120	370	120

<sup>a/</sup> mg/l unless otherwise indicated

surface horizons of the treatment beds contained 1-8 ppm chloride greater than background levels (Table 5). Thus, during five years since the five years since the treatment beds were last excavated, chloride levels in the replacement soil had increased 2.5-4.4 kg/ha-m. Since annual chloride inputs have far exceeded observed levels, the majority of chlorides have undoubtedly been carried through the soil to groundwater by percolating water. Since chlorides are readily carried through the sand and gravel strata, it is reasonable that chloride concentrations in observation wells should be indicative of the degree that percolate from the treatment site affects groundwater quality. In this regard, water quality of wells 1 and 13 appeared unaffected while that of other wells was influenced. Well 12 appeared only slightly affected most the year, but impacts increased during later portion of hydrologic year when groundwater gradients toward well 12 changed. Water quality in wells 2, 9 and 14 were moderately affected while water quality of wells 3, 7, 8, 9 and 10 were more strongly so.

It is uncertain at this time why chloride levels observed directly beneath the treatment site (well 2) averaged less than peripheral wells. One explanation would be that native groundwater has mixed with the percolate.

Sulfate levels in groundwater also point out the relationship of percolate impact on groundwater quality. Effluent sulfate concentrations ranged from 27-74 mg/l with a mean of 42 mg/l. Annual sulfate inputs totaled about 12,000 kg/ha which was not accounted for in sulfate concentration of soil samples. Sulfates in wells 1, 12, and 13 were 7-9 mg/l, while other observation wells had substantially larger levels.

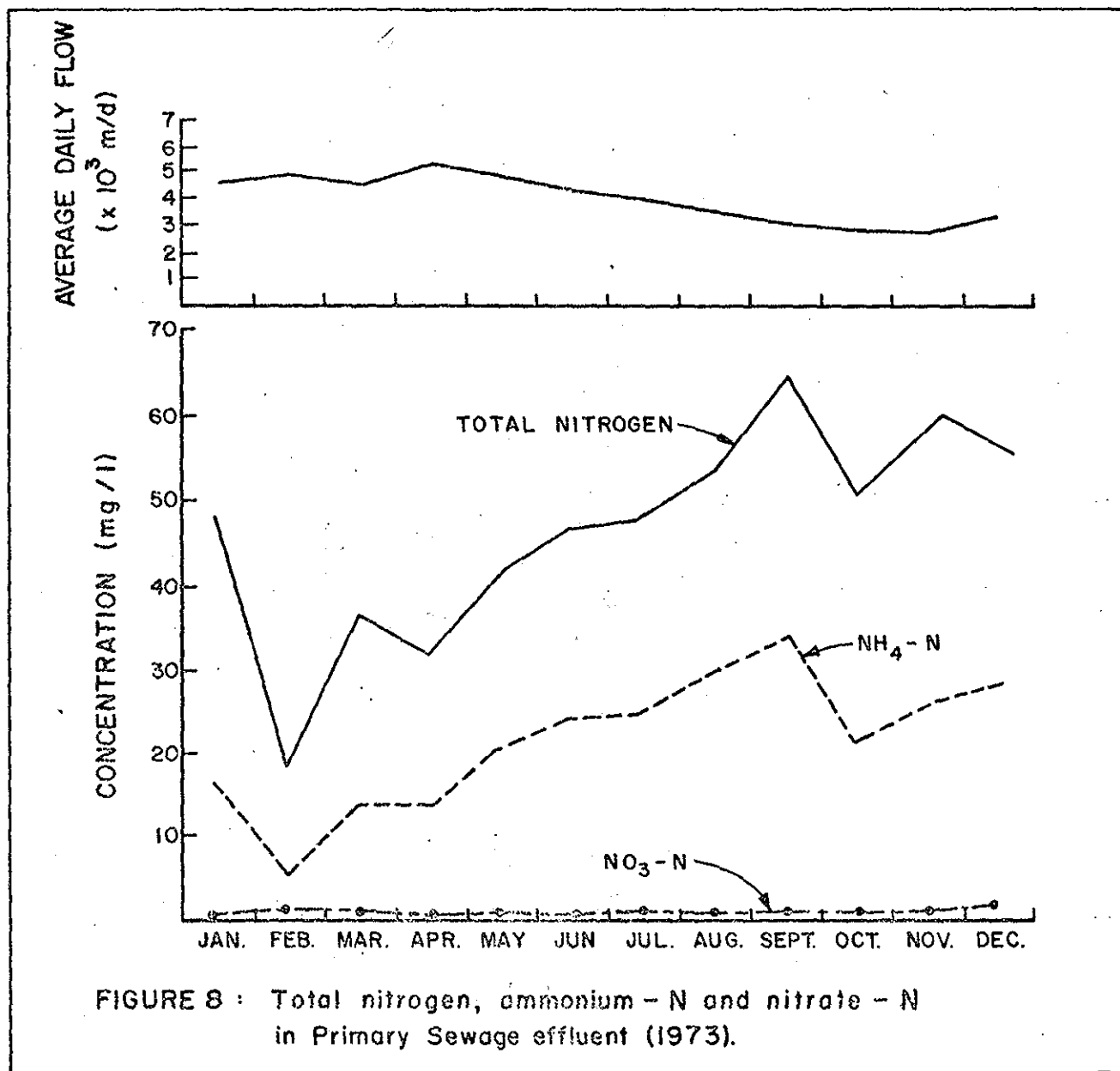
Chemical oxygen demand (COD) of the primary effluent varied seasonally, 110-450 mg/l with a mean value of 192 mg/l. Samples taken from wells 9, 10, 11, 12 and 14 showed COD levels slightly less than those observed for native groundwater, 8-10 mg/l. Wells most impacted by percolate from the treatment area had CODs higher than native groundwater (0.2-3.5 mg/l). Observed levels in wells 2, 3, 7 and 8 ranged from 14-26 mg/l or about 8-14 per cent of mean effluent levels. Five day biochemical oxygen demand (BOD<sub>5</sub>) levels for the primary effluent ranged from 30-185 mg/l with mean of 112 mg/l. Although effluent BOD<sub>5</sub> was high, BOD<sub>5</sub> in groundwater surrounding the application site showed BOD<sub>5</sub> had been effectively removed. Native groundwater BOD<sub>5</sub> levels varied 0.2-3.5 mg/l with a mean of 1.0 mg/l. Levels in the groundwater directly beneath the application area (well 2) ranged 8-18 mg/l with mean of 12 mg/l, however; these levels decreased as the percolate moved away from the treatment site. Groundwater BOD<sub>5</sub> in wells 9, 10, 11, 12 and 14

located 80-145 m from the treatment area were generally less than 1.5 mg/l while average BOD<sub>5</sub> in wells 3, 7 and 8 were less than 2.5 mg/l. Comparisons between the effluent and groundwater BOD<sub>5</sub> levels showed the groundwater levels were generally less than two per cent of the mean effluent concentration.

Calculation of total additions of organic constituents, expressed as BOD<sub>5</sub> and COD, showed approximately 31,000 kg/ha-yr BOD<sub>5</sub> and 53,000 kg/ha-yr COD were added to the treatment beds. These large additions, suggest that organic matter may accumulate in sufficient amounts to reduce water movement into and through the treatment beds. Analysis of soils samples from treatment beds showed that during five years since the beds were last "cleaned," annual accumulation increase of organic matter has averaged about 0.2 per cent per year. This accumulation also included annual additions of the plant material grown naturally on the treatment beds. Although organic matter additions from plants growing on the beds has not been quantified, the relatively small per cent increase of organic matter indicates biological degradation of organic material has been quite effective. Levels of BOD<sub>5</sub> and COD in groundwater further bear this out. Groundwater BOD<sub>5</sub> and COD levels directly beneath and surrounding the treatment site were quite low and in those wells most impacted by percolate COD and BOD<sub>5</sub> were slightly above native groundwater levels.

Total effluent nitrogen which varied seasonally 19-78 mg/l and appeared to be equally organic nitrogen and NH<sub>4</sub>-N with small amounts of NO<sub>3</sub>-N and NO<sub>2</sub>-N, 1.3 and 0.02 mg/l respectively. Total nitrogen in the effluent was inversely related to wastewater flows; low flows, which occurred in late summer through early fall, had high nitrogen levels while high flows during late winter through spring contained smaller nitrogen levels (Figure 8). Seasonal variations were believed to reflect dilution of wastewater by groundwater infiltration into sewer lines and inflow of surface runoff through manholes which occasionally become inundated. Calculations show total nitrogen additions in 1973 were about 13,000 kg/ha.

Comparing nitrogen levels in effluent with those observed in groundwater showed groundwater nitrogen levels were 21-43 per cent of effluent levels. Although groundwater nitrogen levels were substantially less than effluent concentrations observed, levels were greater than average native groundwater level, 2.0 mg/l. Nitrogen in groundwater directly beneath the treatment area was about 14.5 mg/l while wells which are peripheral to the treatment site contained 10-28 mg/l total nitrogen. Wells 9, 12 and 14 positioned about 85 m distance from the treatment beds



had total nitrogen levels averaging about 10 mg/l. Wells 3, 7, 8, 10 and 11 contained the highest total nitrogen levels observed, 15-28 mg/l. Nitrogen levels detected in well 7 were believed to reflect additional effects of leachate from adjacent sludge dewatering beds on groundwater quality.

Organic nitrogen ranged 11-33 mg/l in the effluent, whereas concentrations in groundwater (wells 3, 7, 8 and 9) were only 2-3 mg/l greater than native groundwater levels, 1-2 mg/l. Effluent  $\text{NH}_4\text{-N}$  levels in wells 2, 7 and 8 were generally 5 mg/l. Wells 3, 9, 10, 11 and 12 had less than 4 mg/l  $\text{NH}_4\text{-N}$  and native groundwater about 0.3 mg/l  $\text{NH}_4\text{-N}$ .

$\text{NO}_3\text{-N}$  comprised the major portion of total nitrogen in groundwater of wells affected by percolate from the infiltration area and sludge beds (Figure 9).  $\text{NO}_3\text{-N}$  in wells 3, 7, 8, 10 and 11 varied 10-28 mg/l, although seasonal fluctuations were observed. Highest  $\text{NO}_3\text{-N}$  levels occurred during March through May, and low levels during October and November (Figure 10). Peaks in  $\text{NO}_3\text{-N}$  curves during March and May reflected microbial degradation and nitrification processes of nitrogenous organics due to warmer temperature.  $\text{NO}_2\text{-N}$  was generally less than 0.5 mg/l in most wells although concentrations of 0.5-1.0 mg/l were occasionally observed in wells 3, 7 and 8.

Changes in effluent nitrogen after application and percolation through 18 m of stratified sands and gravels and resultant nitrogen levels in groundwater, provide indirect evidence that nitrification had occurred. Effluent nitrogen was primarily organic nitrogen and  $\text{NH}_4\text{-N}$  with small amounts of  $\text{NO}_3\text{-N}$  and  $\text{NO}_2\text{-N}$  but nitrogen in the groundwater was predominantly  $\text{NO}_3\text{-N}$  with small quantities of organic nitrogen and  $\text{NH}_4\text{-N}$ .

Several mechanisms for nitrogen removal were considered in view of groundwater nitrogen concentration compared to effluent levels. Mixing of the percolate with native groundwater, biological assimilation, soil adsorption, and denitrification could each account for some nitrogen reduction. Dilution of percolate from the treatment site with native groundwater is possible, however, this is not substantiated by the chloride and sulfate levels and concentration of nitrogen observed in wells 3, 4, 5 and 6. These wells were positioned at increasing 1.5 m (5 ft) increments to determine water quality changes through 6.1 m (20 ft) depth. Chlorides in well 3 averaged 257 mg/l while chlorides in wells 4, 5 and 6 were about 140 mg/l. Sulfate levels were similar for all wells (Table 10). Comparison of chlorides and sulfate levels indicate water quality was comparable to levels observed in effluent. This suggests little or no mixing between the percolate and native groundwater. Present information shows saturated thickness beneath the beds (well 2) to be about 11.4 m

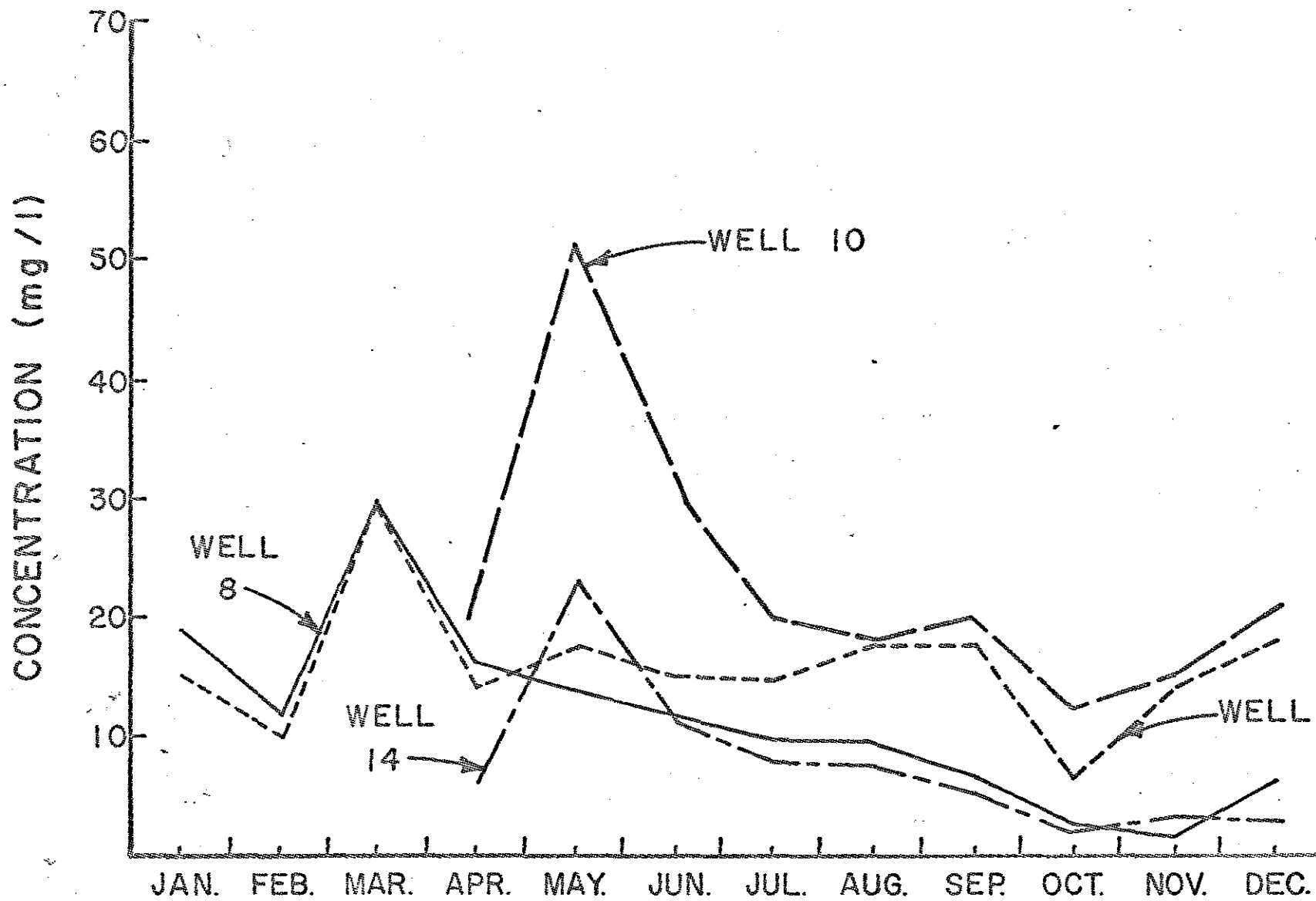


FIGURE 10: Groundwater Nitrate-N levels in selected wells. (1973).

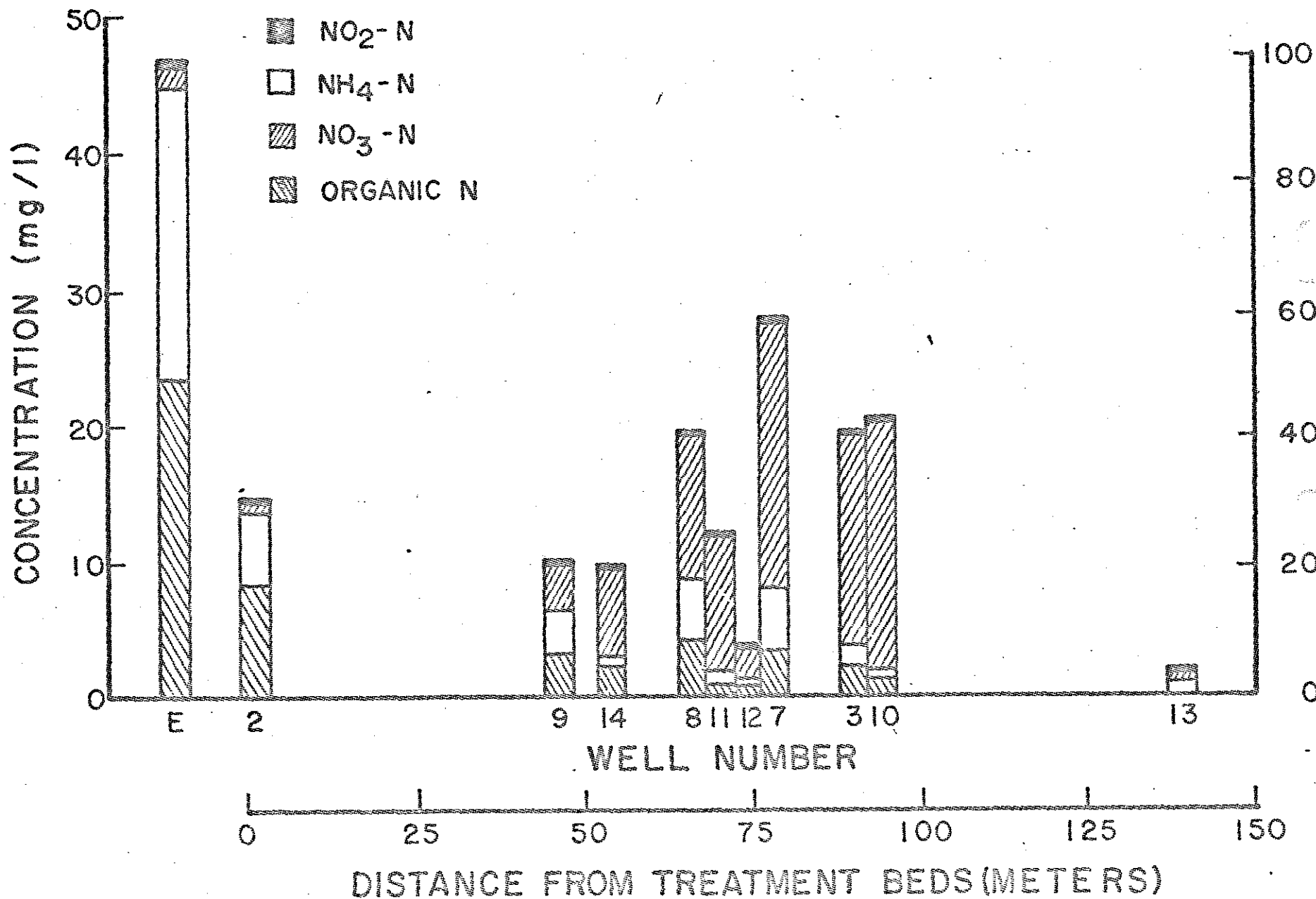


FIGURE 9: Organic-nitrogen, ammonium - N, nitrate - N, and nitrite - N

Table 10

Chemical Characteristic of Primary Effluent Compared to  
Water Quality in Wells 3, 4, 5, and 6 (Average Values)

Parameter <sup>a/</sup>	Effluent	Well			
		3	4	5	6
pH (Standard Units)	6.2-8.0	6.3	6.9	7.0	7.1
Electrical Conductivity (umhos/cm)	511	360	441	502	466
BOD <sub>5</sub>	122	2.5	1.5	1.6	1.9
COD	192	19	29	28	34
Total Nitrogen	47	19.5	20.6	22.1	22.2
Organic-nitrogen	23	2.8	7.9	11	8.0
NH <sub>4</sub> -N	21	1.3	6.4	7	8.4
NO <sub>3</sub> -N	1.3	15.6	5.9	3.8	5.7
NO <sub>2</sub> -N	0.02	0.3	0.4	0.3	0.1
Total PO <sub>4</sub> -P	11	0.9	0.9	0.8	1.0
Ortho PO <sub>4</sub> -P	9	0.2	0.3	0.1	0.2
Cl	150	257	157	159	155
SO <sub>4</sub>	42	46	43	47	46

<sup>a/</sup> mg/l unless otherwise noted

(37.5 ft). Since saturated thickness in well 12 is about 4.2 m (14 ft) and refusal slopes toward the river beneath the treatment beds, native groundwater is probably moving beneath the treatment site at an elevation below that monitored by wells 3, 4, 5 and 6. Chlorides and other parameters determined for wells 3, 4, 5 and 6 were comparable thus mixing does not appear an important consideration for observed differences in nitrogen. Mixing, which may occur, would probably be very small due to short travel distance to wells 3, 4, 5 and 6 plus the fact that mixing in sand and gravel strata would be constrained by the soil particles. Better definition of percolate dispersion will require further inquiry.

Nitrogen reduction from dilution by precipitation could create low nitrogen concentrations for short periods, but gains from precipitation inputs were calculated to be only two per cent of monthly net water budget. This small percentage could not account for differences in nitrogen levels between effluent and observation wells. Dilution would most probably be of short duration, which was not apparent from  $\text{NO}_3\text{-N}$  data plotted in Figure 10.

Biological assimilation of the nitrogen by bacteria and other micro-organisms accounts for some nitrogen removal. Uptake of nitrogen by grasses and other surface macrophytes growing on the filter beds removed about 0.4 per cent of nitrogen applied annually. However, because surface plants and microbes were not removed periodically, assimilated nitrogen is released back into the percolate at a later time through degradation of nitrogenous organic compounds. Slowly, but eventually all nitrogen would be released back to the system.

Fixation of nitrogen in relatively stable organic matter with subsequent adsorption of  $\text{NH}_4\text{-N}$  could account for substantial nitrogen removal (12). Nitrogen in milligrams per gram of soil in the surface samples of filter beds 13, 14 and 15, after receiving primary effluent for five years, were 0.06 to 0.4 more than background levels. Assuming primary effluent quality since October 1968 was comparable to that observed during 1973, annual nitrogen accumulation was 83-134 kg/ha. This represents only 0.6-1.0 per cent of the total annual nitrogen additions.

Adsorption of the  $\text{NH}_4\text{-N}$  on soil exchange sites could remove the  $\text{NH}_4\text{-N}$  applied in the effluent over a three-day period. Following the methodology outlined by Lance (12), ammonium retained in the one meter aerobic surface horizon was calculated. The concentrations of calcium and magnesium, the major competing divalent cation in the primary effluent, were approximately 205. and 4.84 mg/l, respectively.

Ammonia adsorption ratio (AAR) can be calculated using the equation:

$$AAR = NH_4 / (1/2 Ca + 1/2 Mg)^{1/2} \quad (1)$$

where concentrations expressed in meq/l. Experimental data is not available for  $NH_4$ -N adsorption but has been obtained for  $K^+$  adsorption in western soils (24). Since  $K^+$  and  $NH_4$ -N behave similarly in cation exchange reactions, the following equation relating the potassium adsorption ratio (PAR) of a solution to the exchangeable potassium percentage (EPP) of the soil can be used for an estimate of exchangeable ammonium percentage (EAP):

$$EPP = \frac{100 (0.0360 + 0.1051 PAR)}{1 + (0.0360 + 0.1051 PAR)} \quad (2)$$

Knowing AAR (1.41), exchangeable ammonia percentage was estimated using equation 2. For conditions present here, 15.6 per cent of total exchangeable sites in the soil could be occupied by  $NH_4$ -N after effluent of the above composition equilibrates with the soil. Soil which had a mean cation exchange capacity of 1.22 meq/100 gram soil (Table 3) could contain about 0.034 mg  $NH_4$ -N per gram of soil according to these calculations. Thus, one hectare meter of soil could adsorb  $NH_4$ -N in 1.47 m<sup>3</sup> (388 gallons) of primary effluent containing 21.4 mg/l  $NH_4$ -N and the same concentration of calcium and magnesium used above. This would be the amount of  $NH_4^+$  applied to three, 0.32 ha beds over a three-day period where effluent flow was 5061 m<sup>3</sup>.

In order for the ammonium adsorptive capacity to be rejuvenated, adsorbed ammonia must be removed at some time prior to the next application period. Lance and Whisler (12) observed in laboratory column studies that ammonia retained on the soil exchange complex in filter columns would be nitrified during the dry period where aerobic conditions existed. If  $NH_4^+$  adsorption capacity was not rejuvenated, then additional ammonia applied during the next application period would be carried through the filter beds into the groundwater (7).

Ammonium retained on the exchange capacity would nitrify during the ensuing recovery period and create a high concentration of nitrate in water held in the soil. During the initial segment of the next inundation period, the  $NO_3$ -N would be leached through the sand and gravel material which would create nitrate peak in the groundwater. No such peaks were observed during 1973 which probably resulted from the rotation cycle of filter bed inundation which would attenuate any nitrate peak. Increasing the sampling frequency could possibly result in the detection of possible nitrate peaks.

It is uncertain whether nitrogen levels observed in the groundwater were the result of denitrification. However, in view of the large annual nitrogen inputs over the past 30 years and limited capacity of other nitrogen removal mechanisms observed, nitrogen reduction could only be accomplished through the denitrification process.

Various nitrogen forms in groundwater are proof that nitrification of organic and ammonium nitrogen has occurred. The concentrations of organic and ammonia nitrogen in groundwater samples were about 17 and 20 per cent respectively of the effluent levels, while  $\text{NO}_3\text{-N}$  levels increased 8 to 15-fold in groundwater in wells.

Not only must nitrogen be in the proper form ( $\text{NO}_3\text{-N}$ ) for denitrification to occur, but an ample available source of organic carbon must be present for both nitrification and denitrification. St. Amant and McCarty (25) determined 1.3 mg of carbon supplied as methanol is required for every mg of nitrogen reduced to nitrogen gas. Using the equation:

$$\frac{\text{BOD}}{\text{TOC}} = \frac{32}{12} (0.9) (0.77) = 1.85 \quad (3)$$

amount of total organic carbon (TOC) in the primary sewage effluent was approximated where ultimate BOD equaled 90 per cent of the theoretical oxygen demand and five-day BOD is 77 per cent of the ultimate BOD for domestic wastes (9). The C:N ratio was calculated to range 0.85-1.29. The actual ratio may be higher than calculated since about half of the nitrogen was in organic form plus the fact that grasses, algae and other plants growing on the treatment beds, added organic material which is not removed annually and would add to the supply of available carbon.

These calculations show sufficient organic carbon is present at the surface of the treatment beds to facilitate denitrification; provided other environmental conditions are amenable. The high nitrate in the groundwater shows optimum conditions for denitrification did not entirely exist. Findings at the Flushing Meadows Project (7) suggest the two days of treatment bed inundation practiced at Ft. Devens was adequate for nitrification, but is unsatisfactory in duration to enhance denitrification. Lance and Whisler (13) found two days of inundation followed by five days of recovery period was ideal for nitrifying organic and ammonical nitrogen but no net removal nitrogen in the groundwater. Increasing the inundation period to nine days followed by five days recovery, resulted in substantial nitrogen reduction which they attributed to denitrification. Since sufficient organic carbon appears available and nitrification process is operating, adjustments in the inundation period could result in lower

nitrogen levels in the groundwater.

Phosphorus in the primary sewage effluent varied from 6-16 mg/l with an average concentration of 11 mg/l. Calculating phosphorus additions to the treatment site showed substantial amounts of phosphorus was applied, yet groundwater levels remain only slightly higher than background levels. Phosphorus in the native groundwater (well 13) averaged about 0.7 mg/l total  $\text{PO}_4\text{-P}$  of which 0.1 mg/l was ortho  $\text{PO}_4\text{-P}$  or available  $\text{PO}_4\text{-P}$ . Groundwater in the observation wells generally contained less than 2 mg/l total  $\text{PO}_4\text{-P}$  and 0.1-0.4 mg/l ortho  $\text{PO}_4\text{-P}$ . Phosphorus levels beneath the treatment site were 6 mg/l total  $\text{PO}_4\text{-P}$  of which 5.6 mg/l was ortho  $\text{PO}_4\text{-P}$ . However, these levels decreased in the surrounding area to less than 2 mg/l total  $\text{PO}_4\text{-P}$  at a distance of 80-150 meters.

Soil data obtained from treatment beds showed considerable phosphorus was contained in various strata. While soil phosphorus levels varied with depth and between treatment beds, total phosphorus was 5570-7060 kg/ha-m. Thus, phosphorus adsorptive capacity is quite large which would account for the low phosphorus levels in the groundwater, which were about 11 per cent of effluent levels.

Infrequent analysis for fecal coliform bacteria proved negative for all groundwater samples. Total coliform bacteria in the effluent varied  $18 \times 10^6$  to  $53 \times 10^6$  per 100 ml of effluent with a mean coliform level of approximately  $32 \times 10^6/100$  ml. Bacterial analysis of the groundwater samples showed mean total coliform bacteria in well 2 was about 4000/100 ml but was generally less than 200/100 ml in the other wells. Because none of the well casings had been sealed, there was a good possibility that the major portion of the total coliform bacteria observed in the wells were indigenous coliform bacteria and did not originate from the effluent.

Under the present two-day inundation 14-day recovery cycle, impacts of percolate resulting from the rapid infiltration treatment basins at Fort Devens has not substantially impaired groundwater quality in the surrounding area. While concentrations of most water quality parameters were shown to have increased to varying degrees, only  $\text{NO}_3\text{-N}$  and  $\text{NH}_4\text{-N}$  levels were sufficient to cause real concern. BOD, COD, electrical conductivity, phosphorus, sulfate, chlorides and total coliform bacteria were found at somewhat higher levels than observed in native groundwater quality, but not sufficient to warrant major changes in pretreatment of wastewater.

Groundwater nitrogen levels, particularly  $\text{NO}_3\text{-N}$  and  $\text{NH}_4\text{-N}$ , warrant some changes in the operational procedures. While more sophisticated methods could be implemented to reduce groundwater nitrogen levels, nonstructural procedures should be

evaluated, particularly adjustments in the application cycle. Operational changes would be those which would enhance nitrification and denitrification processes so organic nitrogen and  $\text{NH}_4\text{-N}$  would be denitrified to nitrogen gas and volatilized to the atmosphere.

Factors of the nitrification and denitrification processes have been enumerated (1). Because most effluent nitrogen is in the ammonical and organic form, both nitrification and denitrification would have to occur simultaneously in the same treatment bed. Basic differences between the two processes are the form of nitrogen being converted, the oxygen conditions in the micro-environment where the process occurs and the organism necessary to carry out the transformations. Oxidation of organic and ammonical nitrogen to nitrate requires aerobic conditions, while dissimilation of nitrate to nitrogen gas is accomplished under anaerobic conditions. Presently, the application rate and short inundation period followed by a long recovery period is well suited for nitrification. Anaerobic conditions necessary for denitrification are probably not sufficient to optimize nitrate reduction as by nitrate levels in the groundwater. In order to reduce  $\text{NO}_3\text{-N}$  levels in groundwater, the length of inundation period is utilized in the nitrification process (13). However, carbon scarcity should not be a problem at this treatment facility because of the additions of organic carbon in the primary effluent and the plant detritus on the filter beds surface.

Changes in application cycle could result in additional nitrogen removal via nitrification-denitrification but extensive field investigation will be necessary to define the management conditions required at the Fort Devens facility.

Impacts of the percolate on native groundwater has been increased and higher electrical conductivity, nitrogen, chloride and sulfate levels. Although these constituents were found in high concentration, the quality of the groundwater peripheral to the treatment sites continued to meet Public Health Drinking (22) water standards, with the exception of  $\text{NO}_3\text{-N}$  and  $\text{NH}_4\text{-N}$  levels.

Even so, percolate from the rapid infiltration facility at Fort Devens was better than effluent quality reported for tertiary wastewater treatment facilities, which consisted of extended aeration activated sludge with alum polyelectrolyte coagulation, sedimentation, and filtration (16).

## SUMMARY

Investigations conducted at the Ft. Devens, Massachusetts wastewater treatment facility during 1973 have shown rapid infiltration can effectively renovate unchlorinated primary effluent to quality better than conventional tertiary wastewater treatment in a northern environment. Treatment of the 5049 m<sup>3</sup>/day domestic sewage is accomplished using Imhoff tanks and 22 rapid infiltration treatment beds. Effluent loading to the beds was about 27.1 meters. Analysis of soil samples from surface horizons in the 60 m thick deposit of stratified sands and gravels showed substantial increases (200-500 per cent) in the levels of nitrogen, phosphorus, calcium, magnesium, potassium, sodium, and iron over background concentrations. Copper (10-50 ppm) and zinc (20-60 ppm) were also greater than background levels.

Biweekly sampling and analysis of primary sewage effluent and groundwater in fourteen observation wells showed that primary effluent had received renovated comparable to conventional tertiary systems. Groundwater COD and BOD<sub>5</sub> were about 10-20 mg/l and 1.0-2.5 mg/l, respectively. Organic nitrogen and NH<sub>4</sub>-N were generally less than 4 mg/l in the groundwater adjacent to the disposal site while effluent concentrations were 11-33 mg/l and 6.2-42 mg/l, respectively. Effluent NO<sub>3</sub>-N varied 1-2 mg/l during 1973 but was 10-20 mg/l in the groundwater of several wells. Comparing mean total effluent nitrogen with groundwater concentration over the study year revealed groundwater contained only 21-43 per cent of the nitrogen observed in the effluent. However, the greater portion of the 10-20 mg/l total nitrogen remaining was NO<sub>3</sub>-N which were in concentration generally exceeding drinking water standards for municipal waters (19).

Total PO<sub>4</sub>-P in the groundwater was about 2 mg/l of which 0.2 mg/l was ortho PO<sub>4</sub>-P. Total phosphorus in the groundwater was reduced 80 per cent in relation to the levels in the primary effluent. Chloride and sulfate levels were not substantially affected by the filter media.

Infrequent determinations of fecal coliform bacteria in groundwater samples proved negative although substantial numbers were present in the primary effluent. Mean total coliform bacteria levels in the groundwater were about 200/100 ml although the mean number in the effluent was 32 x 10<sup>6</sup>/100 ml.

Results of these investigations show that the treatment beds renovate the primary sewage effluent to levels much better than conventional secondary treatment. Impacts of the percolate to the groundwater quality

adjacent to the disposal site has increased the levels of most constituents. Irregardless, the groundwater adjacent to the site still meets drinking water quality standards with the exception of chloride, nitrate and ammonia levels.

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## APPENDIX A

APPENDIX A  
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Figure 4	Maximum, minimum and average seven day wastewater flow and precipitation data for the Fort Devens sewage treatment facility (1971)
Figure 5	Maximum, minimum and average seven day wastewater flow and precipitation data for the Fort Devens sewage treatment facility (1972)
Figure 6	Maximum, minimum and average seven day wastewater flow and precipitation data for the Fort Devens sewage treatment facility (1973)

## INTRODUCTION

Daily wastewater flows to the treatment facility were obtained from operators logs kept at the treatment facility and by the Facilities Engineer at Fort Devens, Massachusetts. Seven day average flows were the sum of daily flows recorded during a seven-day period divided by seven. Maximum and minimum flows for each seven-day period were the recorded largest and smallest daily flow during the seven-day period.

Climatic data collected at the facility in the past consisted of measuring temperatures once in the morning and again in the afternoon. Precipitation events and wind direction were noted, but measurements were not taken. Precipitation reported in Figures 1 through 6 were obtained from Weather Bureau data for Fitchburg, Massachusetts located 12.9 km ( 8 miles) WSW of the Fort Devens treatment facility. During 1973, precipitation occurring at the treatment site was recorded using a U. S. Weather Bureau rain gauge. Compared to daily precipitation measurement reported for Fitchburg, differences were noted between specific events but monthly and yearly total were approximate.

*To accompany*  
TABLE 1

SIEVE ANALYSIS OF TREATMENT BED SOIL SAMPLES

Sample Depth	Percent of Sample Finer than Sieve Openings of					
(Meters)	2.00 (mm)	1.00 (mm)	0.50 (mm)	0.25 (mm)	0.10 (mm)	0.05 (mm)
<u>Filter Bed 13 a/</u>						
0. -0.15	100	86.0	42.0	10.0	1.0	1.0
0.15-0.46	100	88.0	49.0	17.0	4.0	2.0
0.46-0.61	100	82.0	40.0	14.0	3.0	1.0
0.61-0.91	100	80.0	40.0	16.0	5.0	2.0
0.91-1.52	100	83.0	48.0	26.0	12.0	6.0
1.52-1.98	100	88.0	49.0	21.0	6.0	3.0
1.98-2.59	100	88.0	55.0	37.0	11.0	6.0
2.59-3.05	100	89.0	63.0	41.0	22.0	11.0
<u>Filter Bed 14</u>						
0.0 -0.30	56.0	46.0	28.0	12.0	4.0	1.0
0.30-0.61	58.0	48.0	31.0	10.0	3.0	2.0
0.61-1.22	62.0	46.0	23.0	8.0	4.0	2.0
1.22-1.52	34.0	28.0	19.0	10.0	5.0	3.0
1.52-2.74	63.0	52.0	28.0	12.0	6.0	3.0
2.74-3.05	38.0	30.0	20.0	13.0	7.0	4.0
3.05-4.57	55.0	37.0	20.0	12.0	6.0	3.0
4.57-4.88	98.0	96.0	88.0	25.0	6.0	3.0
4.88-4.97	41.0	34.0	25.0	14.0	7.0	4.0
4.97-5.64	94.0	81.0	44.0	22.0	11.0	6.0
5.64-6.10	77.0	64.0	35.0	16.0	8.0	4.0
<u>Filter Bed 15</u>						
0.0 -0.15	74.0	62.0	38.0	18.0	7.0	3.0
0.15-0.46	40.0	32.0	18.0	7.0	3.0	1.0
0.46-1.22	67.0	50.0	28.0	8.0	2.0	1.0
1.22-1.52	54.0	44.0	28.0	10.0	4.0	2.0
1.52-3.05	73.0	57.0	28.0	12.0	5.0	1.0
3.05-4.27	70.0	54.0	26.0	11.0	5.0	1.0
4.27-4.57	88.0	72.0	43.0	22.0	11.0	6.0
4.57-4.88	72.0	50.0	25.0	12.0	6.0	3.0
4.88-5.79	61.0	48.0	28.0	16.0	8.0	5.0
5.79-6.10	62.0	48.0	29.0	18.0	10.0	6.0

TABLE 1 (Continued)

## SIEVE ANALYSIS OF TREATMENT BED SOIL SAMPLES

Sample Depth	Percent of Sample Finer than Sieve Openings of					
(Meters)	2.00 (mm)	1.00 (mm)	0.50 (mm)	0.25 (mm)	0.10 (mm)	0.05 (mm)
<u>Filter Bed 16</u>						
0.0-0.46	78.0	64.0	36.0	14.0	4.0	1.0
0.46-0.76	52.0	43.0	23.0	9.0	3.0	1.0
0.76-0.91	63.0	53.0	29.0	8.0	3.0	1.0
0.91-1.52	41.0	32.0	19.0	9.0	4.0	2.0
1.52-1.68	88.0	71.0	41.0	19.0	9.0	4.0
1.68-2.13	78.0	56.0	26.0	11.0	5.0	2.0
2.13-3.05	51.0	41.0	22.0	10.0	6.0	3.0
3.05-3.96	48.0	34.0	18.0	8.0	4.0	2.0
3.96-4.57	78.0	51.0	21.0	10.0	4.0	2.0
4.57-6.10	63.0	52.0	33.0	21.0	11.0	6.0
<u>Well #2</u>						
0.0-0.24	82.0	71.0	46.0	18.0	8.0	5.0
0.24-1.28	85.0	76.0	46.0	17.0	8.0	5.0
1.28-1.52	47.0	40.0	24.0	12.0	5.0	3.0
1.52-2.71	74.0	57.0	31.0	11.0	3.0	2.0
2.71-3.05	59.0	46.0	28.0	14.0	7.0	4.0
3.05-4.57	76.0	57.0	29.0	13.0	6.0	4.0
4.57-7.62	68.0	49.0	27.0	15.0	8.0	5.0
6.10-7.62	68.0	49.0	27.0	15.0	8.0	5.0
7.62-9.14	73.0	59.0	36.0	17.0	8.0	4.0
9.14-10.67	90.0	87.0	32.0	8.0	3.0	2.0
10.67-12.19	85.0	76.0	37.0	11.0	5.0	3.0
12.19-13.72	80.0	73.0	23.0	6.0	2.0	1.0
13.72-15.24	84.0	72.0	37.0	12.0	6.0	3.0
15.24-16.74	93.0	84.0	45.0	10.0	3.0	2.0
16.74-18.29	100.0	98.0	83.0	34.0	14.0	5.0
18.29-19.81	100.0	100.0	95.0	54.0	16.0	5.0
19.81-21.34	100.0	100.0	98.0	14.0	1.0	0.0
21.34-22.86	100.0	98.0	74.0	40.0	26.0	5.0
<u>Background Material</u>						
0.0-0.15	91.0	84.0	57.0	28.0	14.0	10.0
0.15-0.30	85.0	78.0	58.0	38.0	22.0	15.0
0.30-0.46	48.0	39.0	30.0	21.0	14.0	12.0
0.46-0.61	71.0	60.0	31.0	6.0	1.0	0.0
0.61-0.91	54.0	45.0	22.0	2.0	1.0	0.0
0.91-1.83	95.0	87.0	62.0	18.0	1.0	0.0
1.83-2.44	88.0	80.0	59.0	27.0	3.0	1.0
2.44-3.05	99.0	99.0	96.0	60.0	4.0	0.0

a/ Particles greater than 2 mm not included in percentage determinations.

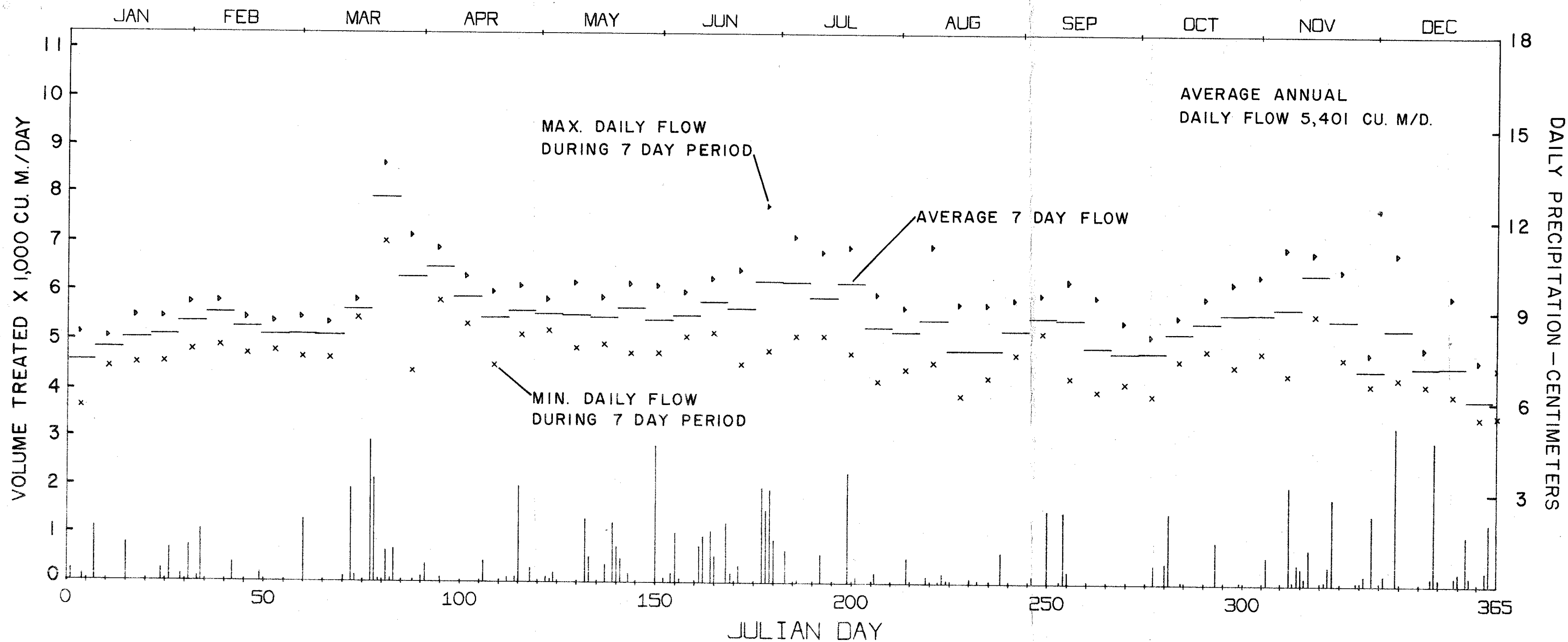


FIGURE 1 : Maximum, minimum and average 7 day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1968)

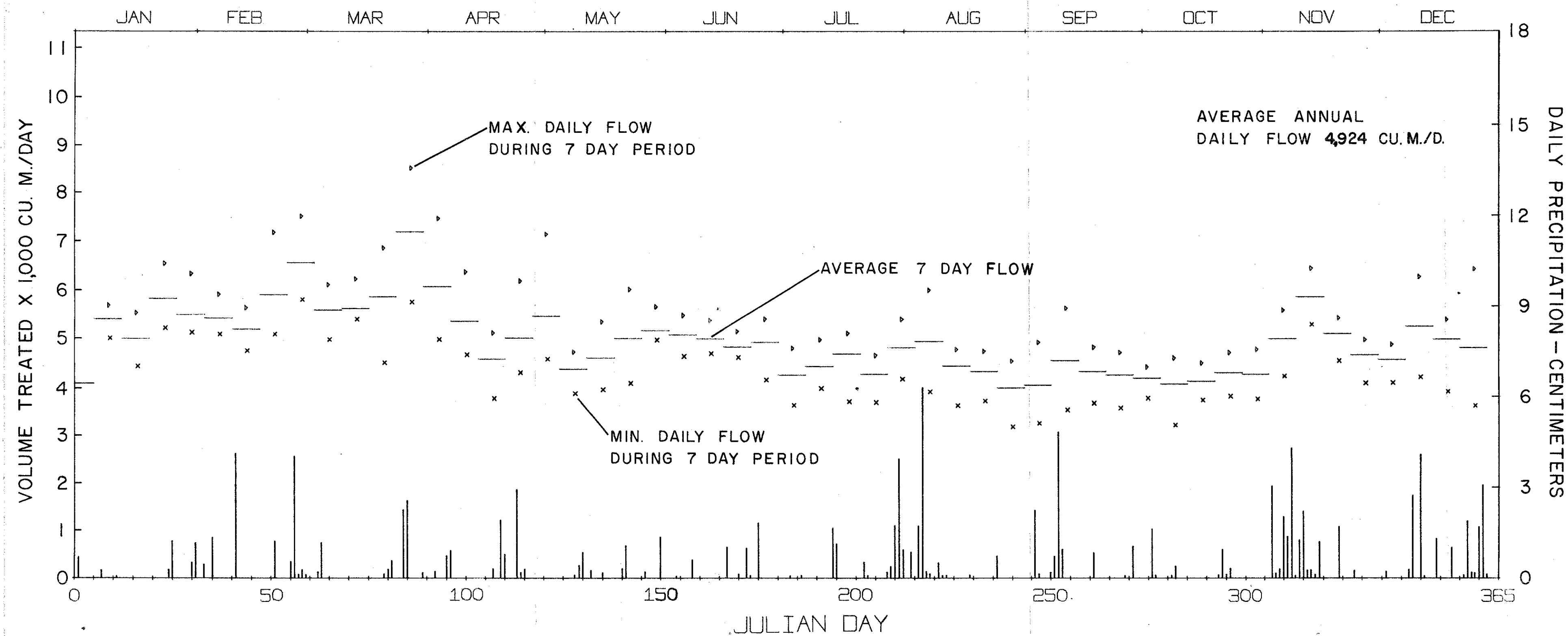


FIGURE 2 : Maximum, minimum and average 7 day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1969).

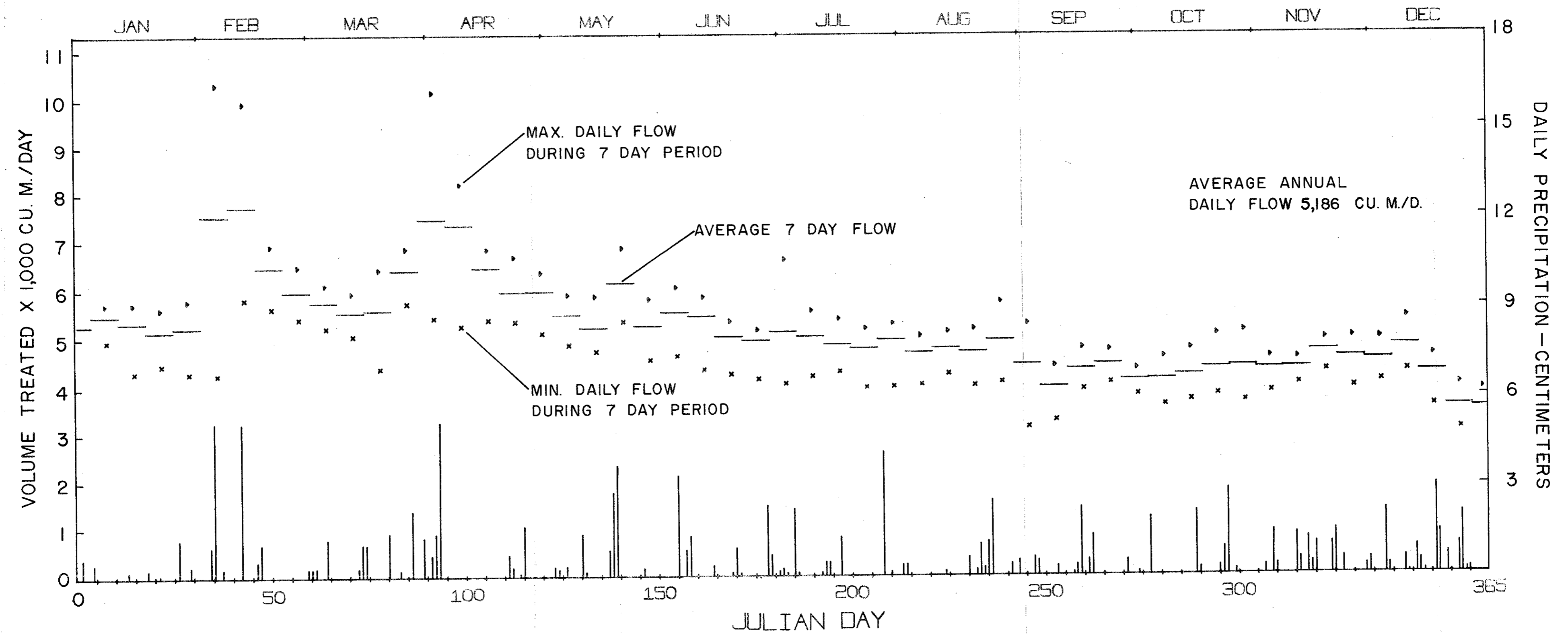


FIGURE 3 : Maximum, minimum and average 7 day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1970).

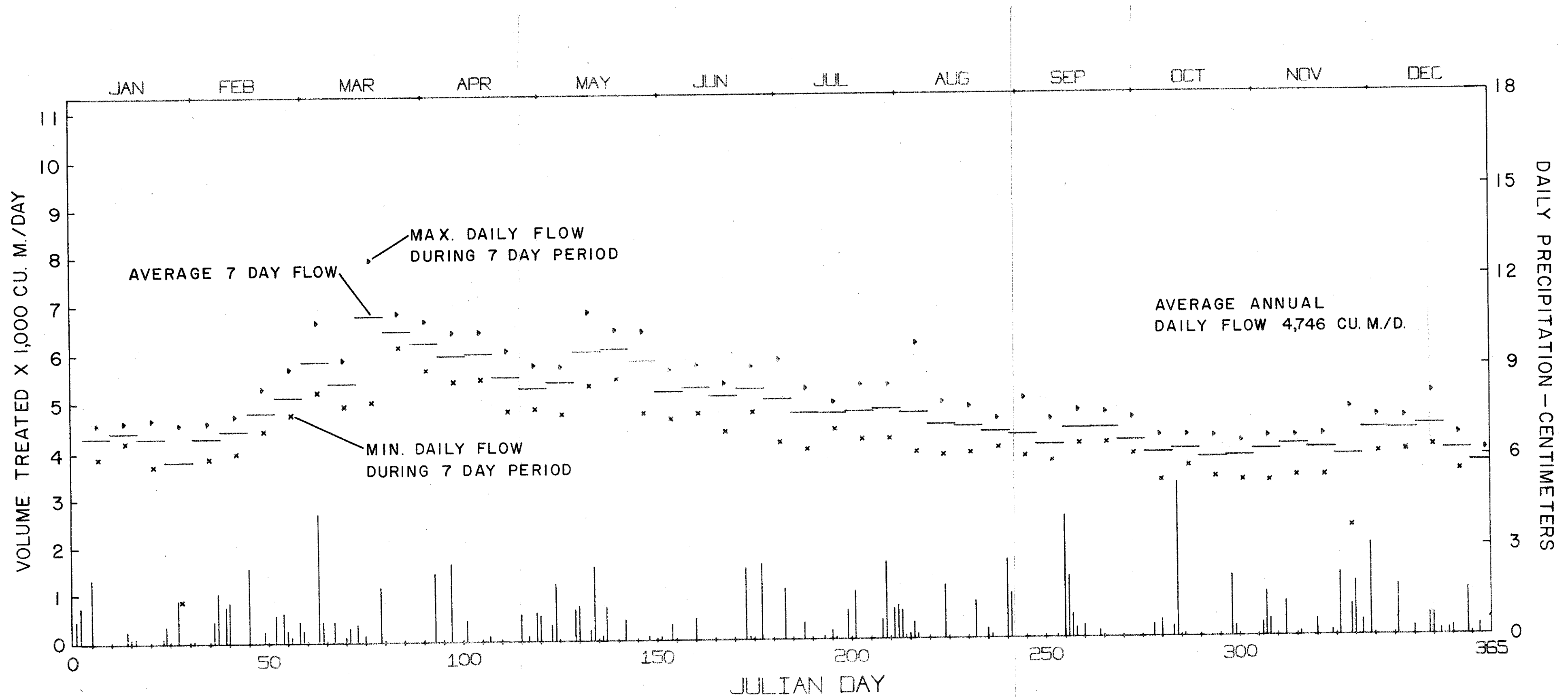


FIGURE 4 : Maximum, minimum and average 7 day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1971).

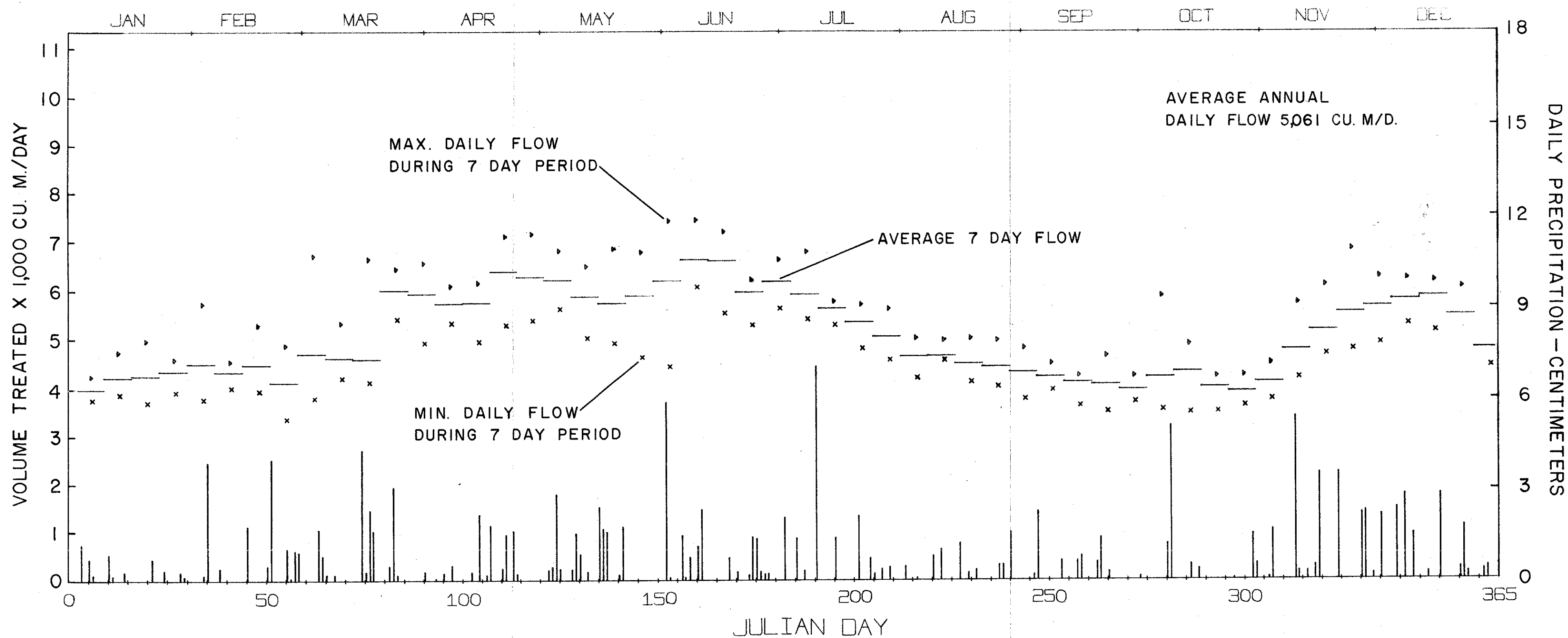


FIGURE 5 : Maximum, minimum and average 7 day wastewater flows and precipitation data for the Fort Devens sewage treatment facility (1972).

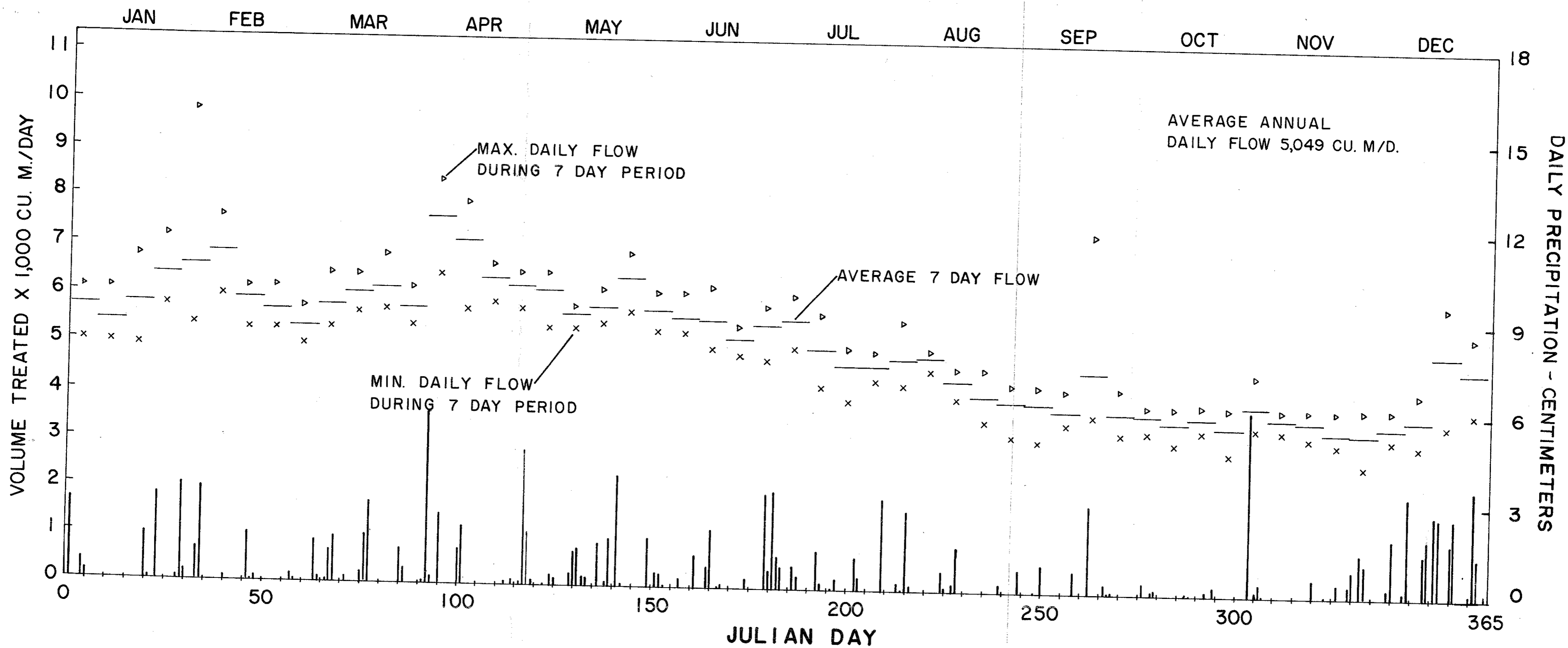


FIGURE 6 : Maximum, minimum and average 7 day wastewater flows and precipitation for the Fort Devens sewage treatment facility (1973).